PB86-178928

Technical Guidance Manual for Performing Waste Load Allocations. Book 4 Lakes and Impoundments. Chapter 2 Nutrient/Euthrophication Impacts

(U.S.) Environmental Protection Agency
Washington, DC

Aug 83

U.S. Department of Commerce
National Technical Information Service
NTIS

50272-101

REPORT DOCUMENTATION PAGE

1. REPORT NO.
EPA 440/4-84-019

2.

3. Recipient's Accession No.
PB86-178928

4. Title and Subtitle
Technical Guidance Manual for Performing Waste Load Allocations
Book IV - Lakes and Impoundments - Chapter 2 - Nutrient/Eutrophication Impacts

5. Report Date
August 1983

6.

7. Author(s)

8. Performing Organization Rept. No.

9. Performing Organization Name and Address
USEPA
Office of Water Regulations and Standards
Monitoring and Data Support Division
Monitoring Branch

10. Project/Task/Work Unit No.

11. Contract(C) or Grant(G) No.
(C)
(G)

12. Sponsoring Organization Name and Address
Same As #9

13. Type of Report & Period Covered

14.

15. Supplementary Notes

16. Abstract (Limit: 200 words)

This chapter is one in a series of manuals whose purpose is to provide technical information and policy guidance for the preparation of Waste Load Allocations (WLAs), which are as technically sound as the current state of the art permits. The objectives of such load allocations are to ensure that quality conditions that protect designated beneficial uses are achieved. An additional benefit of a technically sound WLA is that excessive degrees of treatment, which are neither necessary nor result in corresponding improvements in water quality, can be avoided. This can result in a more effective utilization of available funds.

This chapter addresses Nutrient/Eutrophication impacts in lakes.

17. Document Analysis a. Descriptors

b. Identifiers/Open-Ended Terms

c. COSATI Field/Group

18. Availability Statement
No restriction on distribution. Available from National Technical Information Ser.
Springfield, VA 22161

19. Security Class (This Report)

20. Security Class (This Page)

21. No. of Pages

22. Price

(See ANSI-Z39.18)

See Instructions on Reverse

OPTIONAL FORM 272 (4-77)
(Formerly NTIS-35)
Department of Commerce

PB86-178928

United States
Environmental Protection
Agency

Office of Water Regulations
and Standards
Monitoring and Data Support
Division (WH-553)

August 1983
Final

Water

EPA

Technical Guidance Manual for Performing Waste Load Allocations

Book IV

Lakes and Impoundments

Chapter 2

Nutrient / Euthrophication Impacts

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

Technical Guidance Manual for Performing Waste Load Allocations

Book IV Lakes and Impoundments

Chapter 2 Eutrophication

August 1983
Final Report
for

Office of Water Regulations and Standards
Monitoring and Data Support Division,
Monitoring Branch
U.S. Environmental Protection Agency
401 M Street, S.W. Washington, D.C. 20460

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

AUG 29 1983

MEMORANDUM

SUBJECT: Technical Guidance Manual for Performing Wasteload Allocations
Book IV, Lakes and Impoundments, Chapter 2, Eutrophication

FROM: Steven Schatzow, Director
Office of Water Regulations and Standards (WH-551)

TO: Regional Water Division Directors
Regional Environmental Services Division Directors
Regional Wasteload Allocation Coordinators

Attached, for national use, is the final version of the technical guidance manual for performing wasteload allocations Book IV, Lakes and Impoundments, Chapter 2, Eutrophication. We are sending extra copies of this manual to the Regional Wasteload Allocation Coordinators for distribution to the States to use in conducting wasteload allocations.

Modifications to sections 1, 2, 3.1, and 3.2 of the March 1983 draft include:

- o Increased emphasis on reasons why nitrogen control alone is generally not effective for eutrophication control.
- o Expanding the model selection and use considerations to include the possibility of controlling a nutrient so that it becomes limiting.
- o Listing septic tank and other on-lot disposal discharges as non-point sources.
- o Adding a discussion relating the detail of analysis with the anticipated cost of nutrient removal.
- o Explaining the variability of trophic boundaries and allowing other water quality conditions as a target condition.
- o Including caveats about using nitrogen:phosphorus ratios for determining limiting nutrients.

The remainder of the report is unchanged from the March 1983 draft.

If you have any questions or comments or desire additional information please contact Tim S. Stuart, Chief, Monitoring Branch, Monitoring and Data Support Division (WH-553) on (FTS) 382-7074.

Attachment

ib

TECHNICAL GUIDANCE MANUAL FOR PERFORMING WASTE LOAD ALLOCATIONS

by

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ACKNOWLEDGEMENT

The authors wish to acknowledge the following individuals who contributed in various ways to this chapter.

Jonathan R. Pawlow, the EPA Project Officer, USEPA, Washington, D.C., who provided guidance and direction on the basic content and emphasis, and coordinated input from EPA Regional Offices.

John Mueller of Manhattan College contributed significantly to parts of the chapter associated with simplified eutrophication models of the Vollenweider type. Dr. Robert V. Thomann of Manhattan College also assisted in this effort.

The following kindly provided us with information on mathematical models:

Thomas O. Barnwell, Jr., EPA Environmental Research Laboratory, Athens, GA; Raymond P. Canale, Limno-Tech, Inc., Ann Arbor, Michigan; Steven C. Chapra, Great Lakes Environmental Research Laboratory, Ann Arbor, Michigan; Carl W. Chen, Tetra Tech, Inc., Lafayette, California; John P. St. John, HydroQual, Inc., Mahwah, New Jersey; David P. Larsen, EPA Corvallis Environmental Research Laboratory; G. Fred Lee, Department of Civil Engineering, Colorado State University; Richard A. Park, Center for Ecological Modeling, Rensselaer Polytechnic Institute; William L. Richardson, EPA Large Lakes Research Stations, Grosse Ile, Michigan; R.G. Willey, Hydrologic Engineering Center, Corps of Engineers, Davis, California.

John Marlar, EPA Region IV, Atlanta and the State of Texas Department of Environmental Protection provided insight and information on trophic state criteria which appear to be appropriate in southern and semi-tropic lakes.

Individuals from EPA Headquarters, EPA Regional Offices, and States reviewed and commented on draft versions.

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SECTION 1.0

INTRODUCTION

1.1 PURPOSE

This chapter is one in a series of manuals whose purpose is to provide technical information and policy guidance for the preparation of Waste Load Allocations (WLAs), which are as technically sound as the current state of the art permits. The objectives of such load allocations are to ensure that quality conditions that protect designated beneficial uses are achieved. An additional benefit of a technically sound WLA is that excessive degrees of treatment, which are neither necessary nor result in corresponding improvements in water quality, can be avoided. This can result in a more effective utilization of available funds.

This chapter addresses Nutrient/Eutrophication impacts in lakes.

1.2 RELATION TO OTHER BOOKS AND CHAPTERS

Table 1-1 summarizes the relationship of the various "books" and "chapters" that make up the set of guidance manuals.

These technical chapters should be used in conjunction with the material in Book I, which provides general information applicable to all types of water bodies and to all contaminants that must be addressed by the WLA process.

Table 1-1. ORGANIZATION OF GUIDANCE MANUAL FOR PERFORMANCE OF WASTE LOAD ALLOCATIONS

BOOK I	GENERAL GUIDANCE (Discussion of overall WLA process, procedures, and considerations)
BOOK II	STREAMS AND RIVERS (Specific technical guidance for these water bodies)
	Chapter 1 - BOD/Dissolved Oxygen Impacts 2 - Nutrient/Eutrophication Impacts 3 - Toxic Substances Impacts
BOOK III	ESTUARIES
	Chapter 1 - BOD/Dissolved Oxygen Impacts 2 - Nutrient/Eutrophication Impacts 3 - Toxic Substances Impacts
BOOK IV	LAKES, RESERVOIRS, AND IMPOUNDMENTS
	Chapter 1 - BOD/Dissolved Oxygen Impacts 2 - Nutrient/Eutrophication Impacts 3 - Toxic Substances Impacts

Note: Other water bodies (e.g., groundwaters, bays, and oceans) and other contaminants (coliform bacteria and virus, TDS) may subsequently be incorporated into the manual as the need for comprehensive treatment is determined.

1.3 SCOPE OF THIS CHAPTER

The processes that are significant in eutrophication of lakes are complex, and technical understanding is incomplete from both the qualitative and quantitative standpoints. The expected result and level of confidence associated with waste load allocations addressing eutrophication in lakes will reflect this degree of complexity and incomplete knowledge.

It is not unusual to consider implementation of water pollution control strategies for lakes that are directed toward such relatively qualitative objectives as alterations of a lake's trophic state from eutrophic to mesotrophic. This type of waste load allocation decision has a different basis than a typical dissolved oxygen waste load allocation, which sets a numerical target for dissolved oxygen (for example, five mg/l). Historically, there have been essential differences in the results expected from waste load allocations for control of lake eutrophication and those for dissolved oxygen. When eutrophication is considered, water quality improvements have often been used as the measure of success rather than attainment of specific numerical values of water quality variables.

Waste load allocations for control of eutrophication in lakes are generally designed to reduce nutrient inputs. This strategy presumes that the nutrient to be controlled is a significant factor, limiting the rate of growth and subsequent population of phytoplankton. It further presumes that reducing the population level of phytoplankton will provide the desired control of the complex process of eutrophication and/or eliminate undesirable water quality situations such as algal blooms or low dissolved oxygen concentrations in the hypolimnion. It, therefore, should be recognized by the analyst and by the decision maker that, in general, waste load allocations to control eutrophication in lakes focus directly on nutrient reductions and indirectly on phytoplankton and dissolved oxygen conditions that result from overstimulation by nutrients.

These waste load allocation procedures do not consider ecological factors such as fish populations and species, growth of macrophytes, and species diversity. The quantitative knowledge is not presently available to address water quality problems associated with macrophytes (rooted aquatic plants).

1.4 ORGANIZATION OF THIS CHAPTER

The remainder of this chapter is organized into three parts summarized below.

Section 2.0 discusses the nature of lake eutrophication processes and some factors that influence the performance of a WLA analysis and the interpretation and evaluation of results. This section also identifies and discusses the basic processes that determine the rate and magnitude of the water quality effects, and which are incorporated in the models that will be used to perform a WLA analysis.

Section 3.0 describes and discusses models that can be used to perform WLA's for lake eutrophication. This section covers three classes of models ranging from simplified techniques to complex sophisticated analysis procedures. Example problems are presented to illustrate the use of the simplified models. This section also discusses factors that must be appreciated and given careful consideration in the application of the models and in the interpretation of the resulting calculations.

Section 4.0 provides guidance on the nature and extent of the monitoring programs that may be required for eutrophication analyses. Data requirements for both problem identification and model operation (calibration/verification) are discussed. Some simple statistical procedures for predicting the statistical significance of the data collected from proposed monitoring programs are presented.

SECTION 2.0

BASIC PRINCIPLES

2.1 GENERAL

Two time scales are of interest in the evaluation of lake eutrophication. The first time scale is long and considers the period over which a lake exists and may extend for hundreds or even thousands of years. The second period is the annual cycle when lake chemistry and biology respond to the annual temperature, flow, and solar radiation (light) cycles.

Lakes are considered to undergo a process of "aging" which has been characterized by three qualitatively defined conditions. The initial condition of a lake is termed oligotrophic and is normally associated with deep lakes, where the waters at the bottom of the lake are cold and have relatively high levels of dissolved oxygen throughout the year. The waters and bottom sediments of the lake usually contain only small amounts of organic matter. Productivity in terms of the population levels of phytoplankton, rooted aquatic plants, zooplankton, and fish is usually low. Species diversity is often quite high and chemical water quality is good.

The eutrophic condition of a lake represents the opposite end of the aging process. Eutrophic lakes may be either shallow or deep. They are characterized by high concentrations of suspended organic matter in the water column and by relatively large sediment depths with high organic contents particularly in the upper layers of the sediment. Biological productivity is high and the diversity of biological populations may be somewhat limited.

Coarse (non-game) fish may predominate due to elevated bottom water temperatures and/or depressed water quality. Dissolved oxygen concentrations of bottom waters are usually depressed and in extreme cases of eutrophication may reach zero during summer periods. Generally water quality is low and can result in impairment of beneficial water usages such as water supply, contact recreation, and/or boating.

A third lake condition is mesotrophic which is defined as an intermediate state between oligotrophic and eutrophic. Mesotrophic lakes have intermediate levels of biological productivity and can have some reductions in bottom dissolved oxygen levels. Lakes in this category generally have water quality which is adequate for most beneficial uses but may be deteriorating toward the eutrophic state.

Figure 2-1 is a representation of a transition in lake condition as a function of time. The progression from oligotrophic to eutrophic is shown on the figure as a concentration "C" varying as a function of time. The concentration could represent any of a number of constituents such as phosphorus, chlorophyll, organic carbon, etc. The figure illustrates that there is a general trend of changing -- usually increasing -- concentration as a function of time.

The boundaries between the three stages are not rigidly defined and may vary with regions of the nation and with beneficial uses of lake waters. For lakes in the north temperate zone the following relationship between water quality and lake classification has been suggested. However, it should be

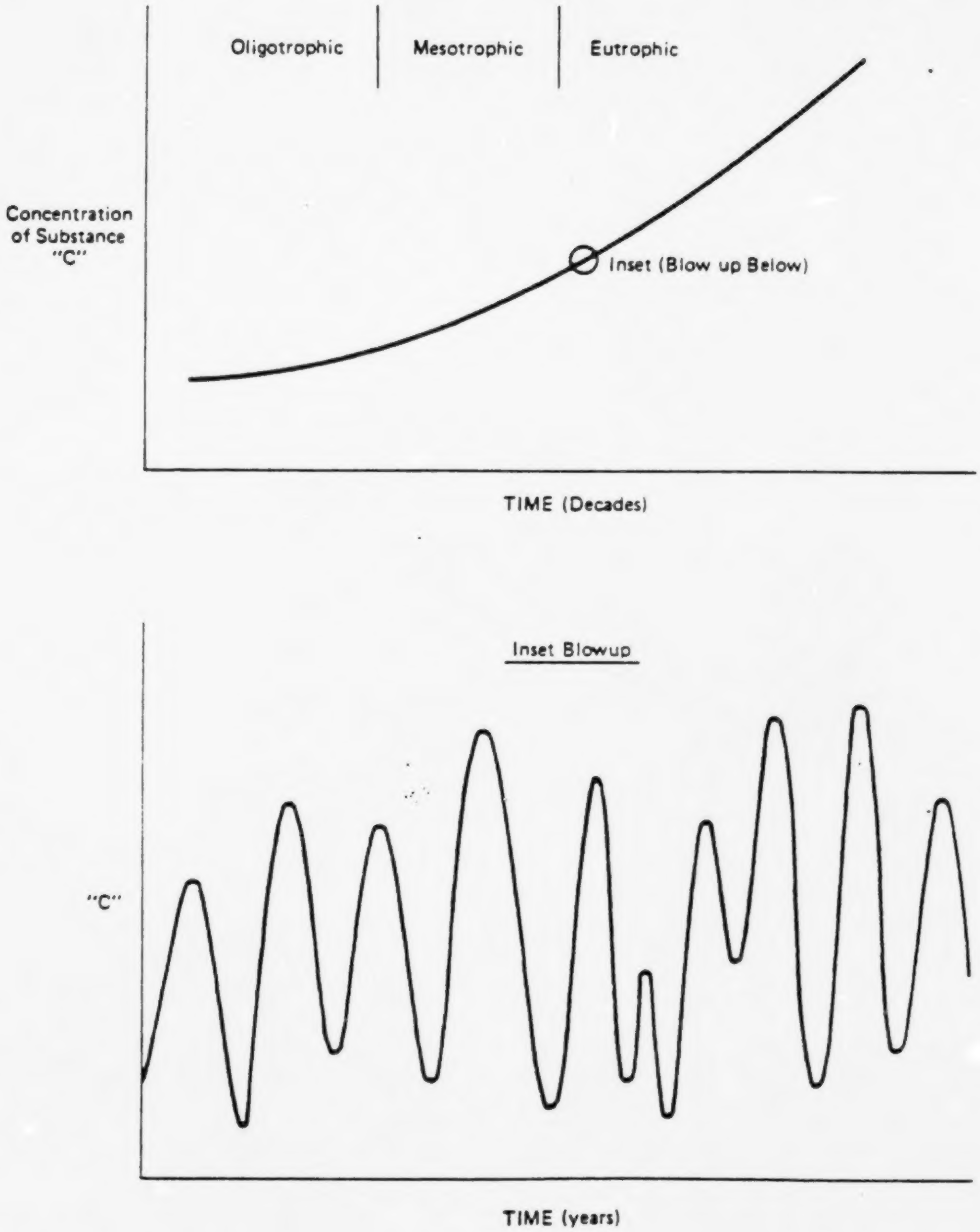


Figure 2-1. Trends in concentration in relation to lake eutrophication.

noted that a number of other trophic classification schemes have been developed (3, 4, 5).

Water Quality Variable	Oligotrophic	Mesotrophic	Eutrophic	Reference
Total phosphorus ($\mu g/l$)	<10	10-20	>20	(1)
Chlorophyll ($\mu g/l$)	<4	4-10	>10	(2)
Secchi depth (m)	>3.7	2-4	<2	(1)
Hypolimnetic oxygen (% saturation)	>80	10-80	<10	(1)

Some lakes in the southern part of the country are often perceived to have higher recreational value where chlorophyll a and total phosphorus levels are substantially higher than those indicated above. As an illustration, the state of Texas employs

Total Phosphorus = 0.4 mg/l
Orthophosphate = 0.2 mg/l
Inorganic Nitrogen = 1.0 mg/l
Chlorophyll a = 50 $\mu g/l$

as alert levels of concern for lake water quality. Lakes and reservoirs in the southwestern region of the country have considered levels of concern ranging from 20 to 40 $\mu g/l$. In addition, low availability of nutrients in areas with high clay content soils entering lakes and reservoirs have resulted in acceptable water quality even with total phosphorus loadings and in-lake concentrations in excess of those indicated above. In summary, judgment is required in establishing target levels of total phosphorus, chlorophyll a or other measures of lake water quality.

As shown in the inset in Figure 2-1 there are often very large short-term yearly variations which characterize the gradually increasing concentration. The large year-to-year variations are related to hydraulic and climatic conditions and may be particularly marked for small to moderate size lakes. Therefore it is often difficult to discern the basic pattern of concentration increases. Further, the large year-to-year variations make it difficult and in some cases impossible to identify the effects of remedial actions such as reductions in nutrient inputs.

The annual cycle in lakes is driven by the seasonal interplay of temperatures, density, and wind. For temperate zone lakes, waters are cold in the winter and may be near the maximum density of water at 39.2°F (4°C). During this period, biological activity is low due to the reduced temperatures and stable density stratification can be present. In spring, diurnal heating and winds tend to promote vertical mixing which results in a spring turnover where water quality tends to be vertically uniform. Light transparancy can be high, and solar radiation and temperature begin to increase. The levels of nutrients available to biological systems are usually elevated as a result of the accumulation during the winter period of low biological activity. These factors combine to yield conditions that can support high levels of growth and biological activity.

As spring yields to summer, the surface waters become progressively warmer and less dense. A stable vertical structure is established which is characterized by a surface layer of uniform temperature and water quality, and an intermediate layer (thermocline) which has significant gradients in temperature and water quality and provides a barrier to vertical transport of dissolved substances such as dissolved oxygen. The bottom waters (hypolimnion) are usually cool with decreasing water quality as the products of active biological productivity in the surface waters settle and begin to accumulate below the thermocline. The late summer is the usual time when high surface chlorophyll and low hypolimnetic dissolved oxygen levels are observed. Nutrient limitations, settling, outflow, and light usually combine to limit growth of phytoplankton. If the dissolved oxygen in the bottom waters drops to zero, release of nutrients from the sediment can be significant. The vertical structure during the summer period will vary from year to year due to differences in solar energy, wind, and flow.

Cooling of surface waters in the fall leads to a uniform vertical structure for both temperature and water quality. Reduced biological productivity is associated with lower water temperatures and reduced solar radiation.

2.2 BASIC PROCESSES

2.2.1 Loads

Nutrient levels in lakes are controlled by external sources to the lake and in-lake processes. External sources of nutrients include municipal and industrial point sources, stream inputs, atmospheric sources, urban drainage, groundwater, agricultural drainage, and other non-point sources surrounding the lake. In-lake processes include sediment release, biological recycling, and nitrogen fixation.

Municipal and industrial point sources may discharge both nitrogen and phosphorus directly into lakes or to streams that eventually drain into lakes. Existing monitoring data should be used or a monitoring network developed to provide reliable estimates of nutrient inputs. Source strength of municipal discharges can vary diurnally and seasonally. Total municipal load usually tends to increase over the years due to population growth.

Stream inputs are the most significant source of nutrients to most lakes. As such, these inputs should be carefully estimated. Estimates can often be obtained by sampling on two to four week intervals and during major storm events. It should be noted that significant increases in nutrient inputs may occur during wet weather flows. Consequently, it is necessary to collect sufficient dry weather flow and concentration data, so as not to overestimate load contributions during dry-weather periods. Nutrient input can then be estimated by multiplying average flow by the flow-weighted concentration, or by a regression equation of phosphorus input on flow. The availability of these nutrients would depend on upstream activities responsible for the nutrient concentrations. Most of the phosphorus, though, is probably not available for immediate uptake.

Atmospheric sources to lakes include precipitation and dry deposition; these sources are frequently considered together as bulk precipitation. Because nutrient forms in precipitation are generally soluble and those in dry deposition generally insoluble, the availability of nutrients from bulk precipitation varies from year-to-year, site-to-site, and storm-to-storm. Nutrient quantities also vary with respect to these parameters. Because nutrient inputs from bulk precipitation are generally small, literature values, despite their limitations, are frequently used for loading estimates.

If literature data indicate bulk precipitation inputs are relatively large, a sampling program may be necessary.

Sampling for nutrient loads from the runoff of urban areas during storm events using automatic samplers can provide estimates of both combined sewer overflow (CSO) and urban runoff inputs. If sampling, which is the most desirable and most costly approach, is not possible, a number of literature sources can be used to estimate inputs (6, 7, 8).

Making reliable estimates of groundwater nutrient input to lakes is difficult. A monitoring system to measure rates of groundwater flow with seepage meters and to determine concentrations of phosphorus in wells has been demonstrated (9). Such a system might be necessary in a lake impacted by septic tank discharges. Because of the spatial and seasonal non-uniformity of groundwater nitrogen and phosphorus concentrations, it is necessary to catalog potential sources of nutrients to surrounding groundwaters. The nutrient forms reaching lakes from groundwater sources would, of course, be soluble and readily available for phytoplankton incorporation.

Agricultural drainage may contribute significantly to the lake nutrient budget. In most cases, agricultural drainage would be estimated as part of the stream input. Estimation of agricultural drainage from lands immediately adjacent to the lake would use the same sampling techniques used for measuring stream input. It may be useful to utilize literature values to initially estimate loads. If these loads are relatively small, a sampling program may not be necessary.

Sediments release nutrients in soluble forms readily available for algal uptake, although the density structure of the lake may hinder immediate uptake. Although this release from sediments is not well understood, laboratory and field investigations have produced numerical estimates of nutrient loss which can be used to compare sediment release to other lake inputs. If the release is relatively high a sampling program may need to be undertaken.

Biological decomposition occurs throughout the water column to make the nutrients locked in organic detritus available for phytoplankton. In addition, phytoplankton and zooplankton secrete and excrete soluble and insoluble nutrient forms.

Nitrogen fixation may be a significant source of nitrogen in lakes with limiting concentrations of nitrogen. During nitrogen fixation, blue-green algae and some macrophytes are able to reduce molecular nitrogen to nitrogen at the ammonia oxidation level. The ability of selected algae and macrophytes to fix nitrogen is frequently cited as one of the reasons phosphorus and not nitrogen is considered to be the limiting nutrient in most lakes.

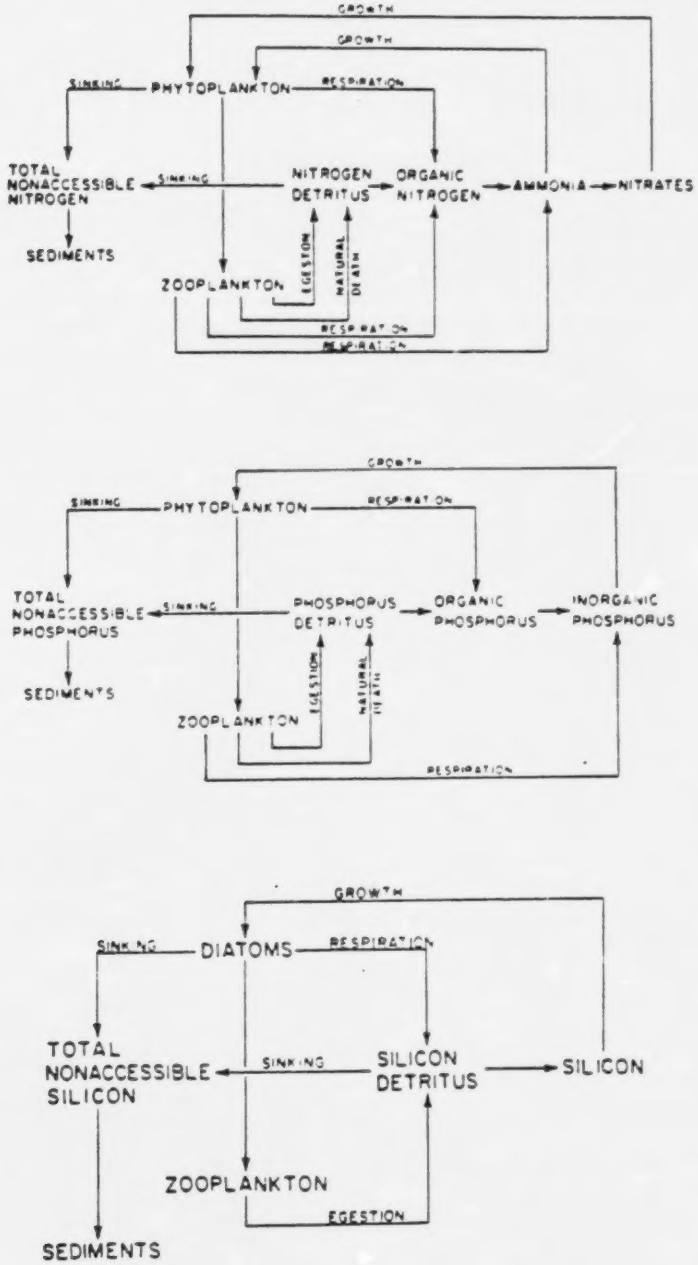
2.2.2 Nutrients

As mentioned above, the two nutrients of greatest concern are nitrogen and phosphorus. In addition to these nutrients, phytoplankton require carbon dioxide and a host of minor elements (potassium, sodium) and trace elements (iron, manganese, cobalt, copper, zinc, boron, and molybdenum) and organic growth factors. Silica is an important nutrient for diatoms, as it forms the basis for their skeletal structure.

Phosphorus in lake inputs and the lake itself can be found in dissolved inorganic and/or organic and particulate forms. Dissolved inorganic forms include the free orthophosphates and the condensed phosphates (pyro, meta, and poly). Orthophosphate is immediately available to phytoplankton growth. Dissolved organic phosphorus includes nucleic acids, nucleotides, and phospholipids, among others. The phosphate part of these molecules must be cleaved by exoenzymes to release phosphorus for uptake. Particulate phosphorus includes algae, bacteria, detritus, and silt, etc. A schematic diagram of phosphorus cycling in lake waters is shown in Figure 2-2.

Analytical testing for phosphorus in water can identify orthophosphate, dissolved and particulate condensed phosphates, and dissolved and particulate organic phosphorus. Total phosphorus is just the sum of all phosphorus species. Levels of total phosphorus in lakes can range from as low as a few μg/l to as high as a few mg/l. These levels are usually reported for elemental phosphorus; in some instances data are reported as phosphates and appropriate conversion is required. Levels of dissolved orthophosphate expressed in terms of elemental phosphorus range from below detection limits to a few hundred μg/l.

Nitrogen can exist in several different forms in lakes and their inputs. Nitrogen in its most reduced state is found in ammonia and various organic nitrogen forms such as purines, pyrimidines, nucleic acids, etc. Ammonia is


GROWTH
GROWTH
SINKING
PHYTOPLANKTON
RESPIRATION
TOTAL NONACCESSIBLE NITROGEN
SINKING
NITROGEN DETRITUS
ORGANIC NITROGEN
AMMONIA
NITRATES
SEDIMENTS
EGESTION
NATURAL DEATH
ZOOPLANKTON
RESPIRATION
RESPIRATION
GROWTH
SINKING
PHYTOPLANKTON
RESPIRATION
TOTAL NONACCESSIBLE PHOSPHORUS
SINKING
PHOSPHORUS DETRITUS
ORGANIC PHOSPHORUS
INORGANIC PHOSPHORUS
SEDIMENTS
EGESTION
NATURAL DEATH
ZOOPLANKTON
RESPIRATION
GROWTH
SINKING
DIATOMS
RESPIRATION
TOTAL NONACCESSIBLE SILICON
SINKING
SILICON DETRITUS
SILICON
ZOOPLANKTON
EGESTION
SEDIMENTS

Source: Canale (10)

Figure 2-2. Nutrient cycle mechanisms for nitrogen, phosphorus and silica in the water column.

immediately available for phytoplankton uptake. Organic nitrogen forms (both dissolved and particulate) may need to be broken down to ammonia for uptake. Some amino acids are immediately available. Other common nitrogen forms include the more oxidized and soluble, nitrate and nitrite. Nitrate is immediately available for uptake but requires the organism to expend more energy to employ this source of nitrogen than for utilization of ammonia.

Measurements of nitrogen compounds are usually reported in terms of elemental nitrogen. In lakes, the sum of the oxidized forms of nitrogen and ammonia may range from 10 μg/l and above. The concentration of organic nitrogen may range up to several mg/l. A schematic diagram of these nitrogen forms as well as their interactions is also shown in Figure 2-2. A pathway for the utilization of silica by diatoms is also shown in Figure 2-2.

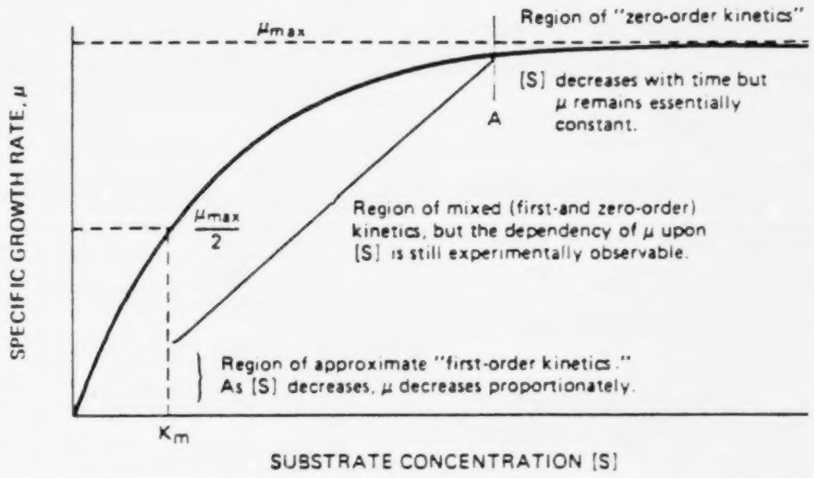
2.2.3 Phytoplankton

The specific growth rate of phytoplankton is controlled by the levels of important nutrients, light and temperature. Overall phytoplankton growth in an area is controlled by this specific growth rate and the effects of death, respiration, settling, zooplankton grazing, and vertical and horizontal transport.

Phytoplankton are one of two main primary producers found in lakes. Primary producers are able to utilize light, carbon dioxide and nutrients to synthesize new organic material. The other primary producers are the rooted or floating aquatic plants (macrophytes). These plants are generally restricted to shallow waters. Phytoplankton are free-floating and transported by currents. In most cases, phytoplankton are more important than are rooted aquatic vegetation in the basic food production of the lake ecosystem, although their relative importance depends on the specific characteristics of the pond/lake in question.

Phytoplankton can be characterized in terms of species, size, composition, growth rates, and pigmentation, among others. Groups of phytoplankton species include diatoms, green algae, nitrogen-fixing blue-green algae and non-nitrogen-fixing blue-green algae. The standing crop of phytoplankton in lakes has been characterized in terms of overall cell counts, individual species counts, mass, and chlorophyll a. These quantities are usually expressed volumetrically, that is, per unit volume. Enumeration and counting can be performed using a microscope equipped with a hemocytometer for nanoplankton counting or with a Sedgwick-Rafter cell for enumeration of larger species. Phytoplankton mass has also been characterized after drying, ashing, and weighing. A problem with this determination is that all biomass in a lake water sample, including bacteria, detritus and zooplankton, will also be measured as phytoplankton. The characteristic algal pigments include chlorophylls (a,b,c), xanthophylls, and carotenes. Chlorophyll a is the pigment typically used to quantify phytoplankton populations. Chlorophyll a is measured either spectrophotometrically or fluorometrically.

The effect of phosphorus on phytoplankton growth is frequently described in terms of an equation attributed to Michaelis-Menton to describe enzyme kinetics. As shown in Figure 2-3, this equation expresses the specific growth rate as a function of phosphorus concentration and two coefficients,



Michaelis-Menton Kinetics

$\mu = \mu_{max} \frac{S}{K_m + S}$ where μ_{max} = Maximum specific growth rate

μ = Specific growth rate

S = Substrate concentration

K_m = Substrate concentration at which $\frac{\mu_{max}}{2} = \mu$

Source: Rich (11)

Figure 2-3. Specific growth rate versus substrate concentration.

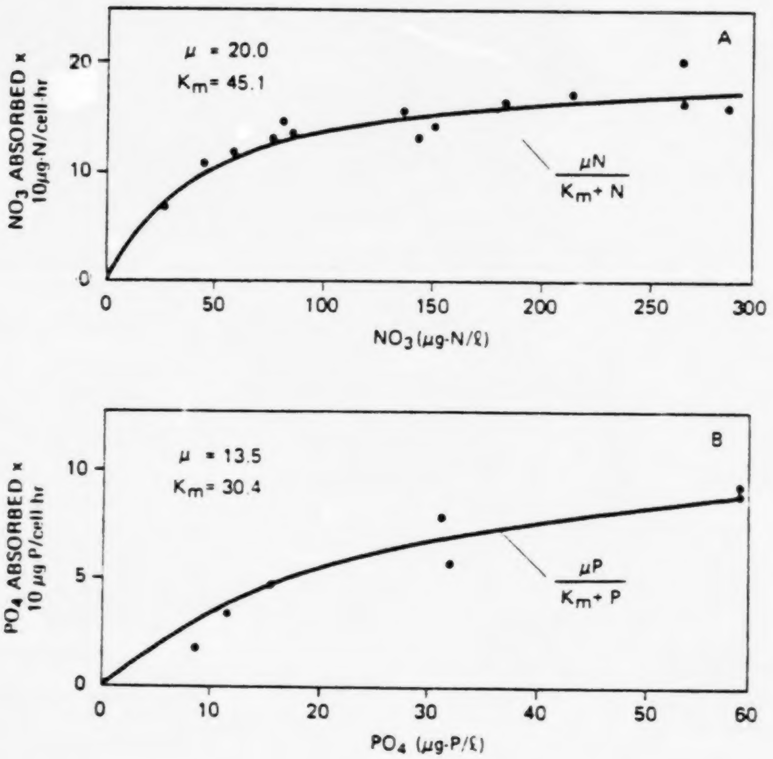
the maximum growth rate coefficient, μ_{max} and the half-saturation coefficient, K_m. A plot illustrating the similarity between nutrient uptake data and the Michaelis-Menton formulation is shown in Figure 2-4. Growth rates of individual phytoplankton have traditionally been measured in terms of mean doubling times. The mean doubling time is simply the natural logarithm of two divided by the maximum specific growth rate.

The growth rate variation of phytoplankton as a function of nitrogen concentrations can also be modeled using the Michaelis-Menton formulation. An example of the close correlation between this formulation and real data is shown in Figure 2-4. Similar correlations have been found when ammonia and various organic nitrogen compounds are the substrate molecules.

Phytoplankton growth rates vary with temperature and light intensity. Examples of the rate of photosynthesis as a function of light intensity are shown in Figure 2-5. Mathematical representations of light dependent growth are discussed in subsequent sections of this report. It can be observed from the data in Figure 2-5 that an optimum light level exists.

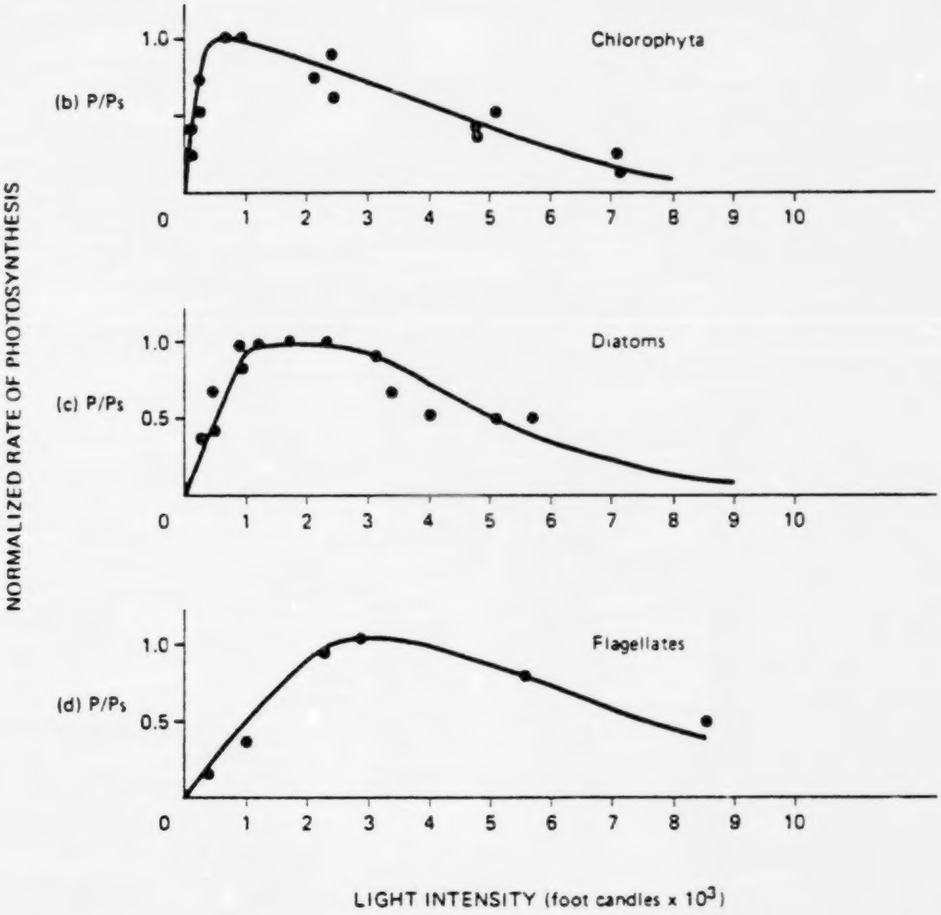
Other processes affecting phytoplankton levels are respiration and death. Respiration is a biochemical process that occurs continuously day and night and results in consumption of some portion of the photosynthetically fixed carbon in the system. Hydrolysis of the phytoplankton cell follows death.

Factors affecting phytoplankton growth in a particular volume include advective and dispersive transport, settling, and zooplankton grazing.



Source Manhattan College (12)

Figure 2-4. Nutrient absorption rate as a function of nutrient concentration: comparison of Michaelis-Menton theoretical curve with data from Ketchum.



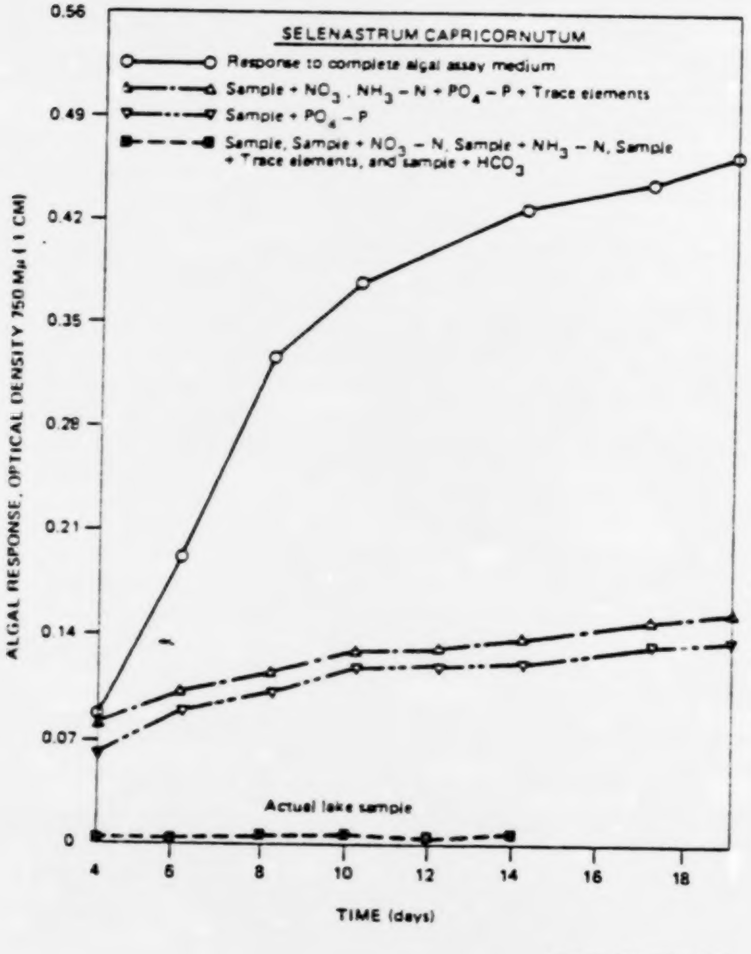
Source: Manhattan College (12)

Figure 2-5. Normalized rate of photosynthesis versus light intensity.

Transport is the gain or loss of plankton from the system as a result of water movement. Settling results in a loss of phytoplankton from the euphotic zone. Settling occurs even though the density difference between phytoplankton and water is very small. Zooplankton grazing provides another check on phytoplankton populations.

As mentioned above, there are a number of factors that can limit phytoplankton growth. For the purposes of modifying the productivity of a lake, it is important to identify those limiting factors which can be controlled. Very little can be done to control the intensities or concentrations of light, temperature and the various trace elements and organic growth factors. Because some control can be exerted over the concentrations of nitrogen and phosphorus in lakes, considerable effort has been exerted to define the effects of these nutrients. While nitrogen contributions from point sources are controllable, the greater solubility of nitrogen compounds in non-point sources makes control difficult. The ability of blue-green algae to fix nitrogen from the atmosphere also reduces the importance of nitrogen control. In addition, control of nitrogen alone may cause a shift from green to blue-green algae, thereby reducing the effectiveness of control. As inorganic phosphorus compounds are much less soluble than inorganic nitrogen compounds and tend to adsorb onto natural surfaces, control of phosphorus point sources can be more effective.

One method to determine the limiting nutrient is the algal bioassay. In this procedure lake water samples are spiked with incremental additions of the nutrient(s) being investigated (see Figure 2-6). A number of samples



Source: Thornton (19)

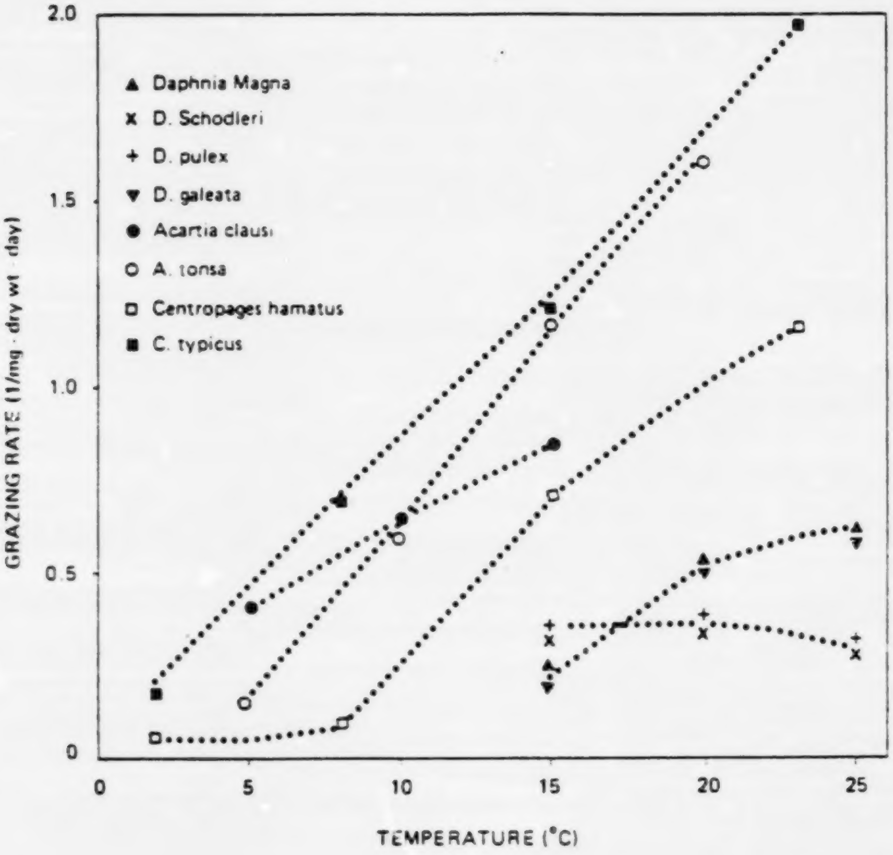
Figure 2-6. Use of algal bioassays to determine the limiting nutrient in stream or lake waters.

with different levels of nutrient(s) are then incubated in the laboratory for a specified period of time under specified conditions (13). These samples may also be incubated in-situ.

Considerable controversy still exists over the role of rooted aquatic plants in eutrophication dynamics. The area of controversy revolves around the origin of nutrients used by these primary producers. Bole and Allan (14) reported levels of phosphorus that control the source (sediment or water) of nutrient uptake by aquatic weeds. In addition macrophyte growth was found to increase in response to steadily increasing nitrogen and phosphorus concentrations in waters of the Goczalkowice River (15). On the other hand, Carignan and Kalf (16) showed that mobile phosphorus in sediments closely matched the phosphorus uptake by macrophytes.

Other investigations have found that aquatic plants were limited only by light and space requirements. For instance, Sheldon and Boylen (17) found that the density of aquatic plants decreased linearly as the depth of the sampling location from the shore increased. Jupp and Spence (18) found that wave action, phytoplankton competition, and grazing by waterfowl had more effect on macrophyte biomass than other factors.

Zooplankton, including protozoa, rotifers and crustacea, form the next link in the lake food chain. These primary macroconsumers provide the link between the primary producers (phytoplankton) and such secondary consumers (carnivores) as predaceous insects and game fish. As such, zooplankton provide a primary constraint on phytoplankton growth. The basic mechanism by which zooplankters feed is by filtering the surrounding water and clearing it



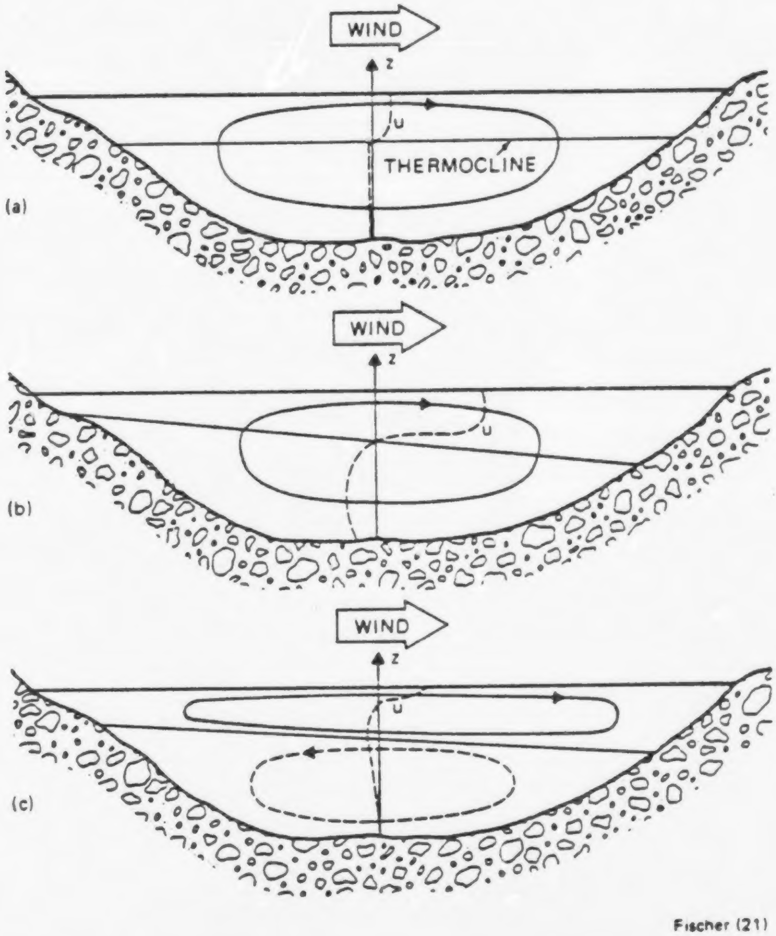
Source Manhattan College (12)

Figure 2-7. Grazing rates of zooplankton versus temperature.

in an inclination in the water surface (wind setup) and a counterflow in the hypolimnion. As illustrated in Figure 2-8, such motions can cause significant horizontal as well as vertical transport in both the epilimnion and the hypolimnion.

Heat input and depth are two important factors determining the thermal structure of lakes. Insolation heats the water and this energy is distributed over an upper mixed layer. During any particular period, the difference between the heat input from the sun and heat losses through back radiation, evaporation, convection, and outflow determines whether the lake surface is heating or cooling. Heating during the spring and summer may lead to stratification in some lakes. Stratification in extremely shallow lakes is not expected because wind-induced turbulence is large enough to mix lake waters to the bottom. In lakes that do stratify, the depth of stratification is determined by the amount of energy causing turbulence, which is related to surface wind stress and the temperature differences between upper and lower layers.

The turbulence found in the epilimnion may be important in keeping phytoplankton and zooplankton in suspension. At the thermocline, which is an area of extreme stability, vertical transport between the epilimnion and the hypolimnion is limited. Internal seiches in the thermocline may cause spatial and temporal variations in the location of the thermocline, but probably do not increase transport across the thermocline.


WIND
z
u
THERMOCLINE
(a)
WIND
z
u
(b)
WIND
z
u
(c)

Fischer (21)

Figure 2-8. Formation of baroclinic motions in a lake exposed to wind stresses at the surface: (a) initiation of motion, (b) position of maximum shear across the thermocline, (c) steady-state baroclinic circulation.

Most calculation frameworks for waste load allocation studies employ simplified representations of the horizontal and vertical transport processes. Most calculation procedures consider the lake as one completely mixed system. Even more complex analysis frameworks employ limited spatial segmentation to represent important horizontal and vertical transport processes.

2.2.5 Bottom Processes

Bottom processes are responsible for the sequestering and recycling of detritus and nutrients. Recycling begins with the consumption of dissolved oxygen in the sediment by heterotrophic bacteria utilizing deposited organic material. As these bacteria consume oxygen, the sediment tends to become anaerobic. Transport of oxygen into the sediment occurs through diffusion and through the mixing of the upper 5 to 20 cm of the sediment because of the activities of benthic organisms, bottom-feeding fish, bubbling of fermentation gases, and wind-induced currents. The vertical migration rate of reduced oxygen-demanding substances up through the sediments and into the overlying waters is also determined by these processes.

Previous research indicates that in some cases a thin, oxidized micro-layer at the sediment-water interface regulates exchange between water and sediments. The depth of the oxidized micro-layer is determined by the dissolved oxygen concentration in the water overlying the sediment and by the rate of oxygen consumption in the sediment. When the oxidized layer is eliminated, an abrupt release of iron and phosphate has been noted. Release

SECTION 3.0

MODEL SELECTION AND USE

3.1 INITIAL CONSIDERATIONS

3.1.1 Limiting Processes

As indicated, waste load allocations rely on the concept of reducing inputs of a limiting nutrient to control growth of phytoplankton or by controlling a nutrient so that it becomes limiting. Other factors also limit the rate of phytoplankton population growth and resultant population levels. Among the other factors which may be important are light limitations, hydraulic retention times, settling, and grazing by zooplankton. In most site-specific situations, several factors combine to limit phytoplankton growth and populations. If the limitations on growth imposed by factors other than nutrient concentrations are large, it may not be economically feasible to control eutrophication with reductions in nutrient inputs. The non-linear modeling procedures discussed elsewhere in this document consider the influence of certain other factors on the growth of phytoplankton while the less complex analysis procedures which are presented assume that nutrients are the significant factor limiting growth.

Phosphorus has been found to be the nutrient which limits growth in many lakes. In addition, a lake that is not presently phosphorus limited could become so if a large percentage of the phosphorus loading to the lake is removed. However, nitrogen and silica have also been identified as limiting nutrients in some lakes and for some periods of time, depending upon the seasonal variations in species and phytoplankton populations. The analyst needs to consider the following questions on a site-specific basis:

might be associated with individual nutrients can be obtained by application of equations (3-1), (3-2), and (3-3).

$$R_p = \frac{C_p}{K_{mp} + C_p} \qquad (3\text{-}1)$$

$$R_n = \frac{C_n}{K_{mn} + C_n} \qquad (3\text{-}2)$$

$$R_s = \frac{C_s}{K_{ms} + C_s} \qquad (3\text{-}3)$$

where

R_p, R_n, R_s, = Approximate reduction in ambient growth rate associated with nutrient limitations from phosphorus, nitrogen, and silica respectively.

C_p = Observed orthophosphate concentration (μg/1)

C_n = Observed inorganic nitrogen concentration ($NH_3 + NO_3 + NO_2$) (μg/1)

C_s = Observed dissolved inorganic silica concentration (μg/1)

K_{mp} = Half saturation constant for phosphorus (μg/1) (reasonable value = 7 μg/1; range from 2 to 50 μg/1)

K_{mn} = Half saturation constant for inorganic nitrogen (μg/1) (reasonable value = 25 μg/1; range from 10 to 400 μg/1)

K_{ms} = Half saturation constant for inorganic silica (μg/1) (reasonable value = 30 μg/1)

The nutrient with the lowest "R" value would be considered the limiting nutrient and the approximate reduction in ambient growth rate associated with that nutrient would be calculated assuming a single nutrient limits growth the phosphorus mineral apatite and refractory organic nitrogen, are considered to be unavailable biologically on the time scales of concern.

There are no laboratory or experimental techniques which can be employed in waste load allocation studies to differentiate readily available, ultimately available, and unavailable forms of any of the nutrients. Several (23, 24) chemical and biological techniques are being developed in ongoing research efforts for differentiating available and unavailable phosphorus. They should not be considered for use in most waste load allocation projects at this time.

Ideally, the load allocations for nutrients should consider the nutrients which influence biological processes. These nutrients are the sum of the readily available and a portion of the ultimately available nutrients. The portion of the ultimately available nutrients included in this idealized situation would vary on a site-specific basis depending on the nutrient cycling processes (particularly bottom processes) and lake detention time. Waste load allocations have not and cannot approach this ideal. Rather, the less complex approaches to eutrophication analysis, both steady-state (Vollenweider) and time variable (residence calculation), generally deal with total nutrient inputs and concentration levels. By contrast, non-linear eutrophication modeling analysis usually employs separate nutrient variables in the calculations for readily available and ultimately available nutrients. The latter variable usually includes ultimately available and unavailable nutrients. The values of the settling, mineralization, and other coefficients in these models are probably altered slightly by inclusion of unavailable where:

SRP = soluble reactive phosphorus

PP_t = total particulate phosphorus

a = factor ranging from .1 to .3

For municipal discharges, it has been estimated (27) that the available phosphorus averages 72 percent after chemical precipitation for phosphorus removal. The available P in effluents averaged 82 and 55 percent for soluble and particulate phosphorus respectively.

Quantitative evaluation of available nutrients in waste load allocations for control of eutrophication in lakes cannot be carried out with the existing base of data and knowledge. Qualitatively, it appears that point source controls for discharge directly to lakes will be most effective in controlling phytoplankton growth. Point source discharges located upstream on tributaries have a lower probable level of effectiveness as do non-point source controls which would impact phosphorus inputs from tributaries. Therefore from the standpoint of availability of nutrients, waste load reductions should initially be directed toward point sources that directly discharge to a lake. If additional reductions in nutrients are required, the point and non-point sources on tributaries should be the next types of sources controlled with distance from the lake providing a qualitative indication of probable relative availability. A more quantitative estimate of tributary loads can be obtained from ambient water quality monitoring of lake inputs in question. The overall nutrient removal required will be determined by the quantity of nutrients associated with each source, the cost of removal and qualitatively by the biological availability.

3.1.3. Estimating Loadings

Loading estimates for nutrient inputs to lakes are required for all of the analysis frameworks available to examine waste load allocations in lakes. The time scale over which mass loading estimates should be developed is determined by the retention time of the lake. Generally, annual loading estimates are required. For small lakes or lakes having short detention times, the annual load may have to be subdivided seasonally. The loading estimates should define mass inputs of the limiting nutrient by type and location of a source. As indicated in Section 2.2.1 and illustrated in Figure 3-1, the type of sources which must be considered are:

1. point sources directly discharging to the lake
2. atmospheric inputs
3. intermittent discharges directly to the lake from CSO's and urban runoff
4. point sources, CSO's, and urban runoff discharges on tributaries
5. non-point sources which enter the lake or tributaries
6. erosion of tributary banks and the lake shore line
7. in-lake sources (such as release of nutrients from bottom sediments).
8. septic tank and other on-lot disposal discharges

At a minimum, the measured nutrient loads should include total nutrient levels (total P, total N, etc.) and readily available nutrient levels (orthophosphate, NH_3, NO_2, NO_3). If possible, measures of particulate associated nutrients should also be obtained.

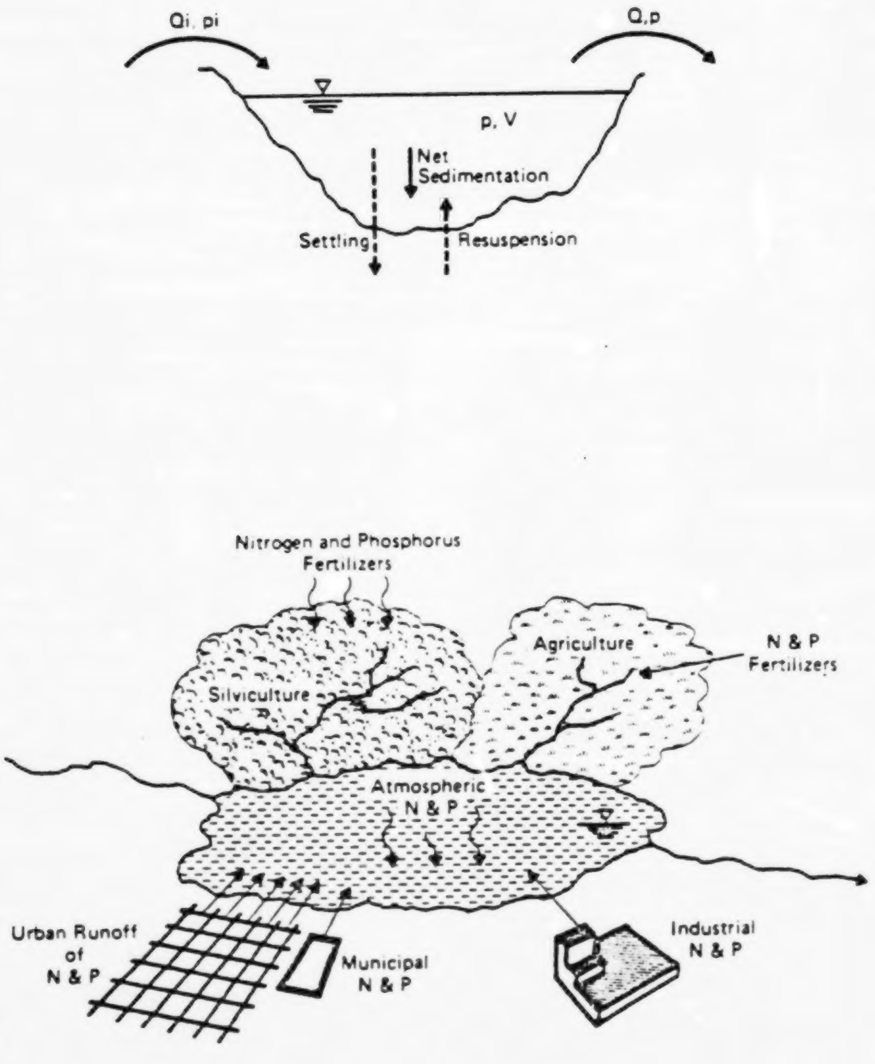

Qi, pi
Q,p
p, V
Net Sedimentation
Settling
Resuspension
Nitrogen and Phosphorus Fertilizers
Agriculture
N & P Fertilizers
Silviculture
Atmospheric N & P
Urban Runoff of N & P
Municipal N & P
Industrial N & P

Figure 3-1. Problem framework and nutrient sources.

3.1.3.1 Point Source--

Generally, routine plant monitoring samples and flow measurements will adequately define the loads from point sources if the limiting nutrient concentration is among the measurements obtained. The length of data record analyzed should include a full year's data, containing any seasonal and weekly variations.

3.1.3.2 Atmospheric Inputs--

Data over one year at several stations on the lake would be ideal. Practically, several site-specific samples obtained to reflect seasonal factors could be combined with data in the literature to develop estimates of atmospheric inputs. Spatial distribution of atmospheric input may be important in larger lakes or those where land use varies significantly around the lake shore.

3.1.3.3 Intermittent Discharges (CSO's and Urban Runoff)--

It will usually be necessary to obtain some representative samples of CSO and urban runoff loads for a site-specific project. It may be necessary to differentiate the soluble and particulate nutrient loads from these source types if treatment feasibility must ultimately be determined.

In almost all cases, mass loading estimates on a seasonal or annual basis will be made by projections from limited sampling in time and space. A model, such as SWMM or Storm, may be employed to project annual loads. Alternatively, estimates of the annual load may be obtained by considering the nutrient concentration as an independent random variable (usually log normally distributed). The natural log of the mean and variance can be estimated from a log normal probability plot. Substituting these into equation (3-4) provides an estimate of the arithmetic mean (maximum likelihood estimate) (U_x) (6) which, when multiplied by the runoff flow for the year, will provide an estimate of the nutrient load.

$$U_n = e^{\left(M_n + S_n^2\right)} \tag{3-4}$$

where

U_n = maximum likelihood estimate of the runoff concentration

M_n = natural log of the concentration at 50% from log probability plot

S_n^2 = natural log of the variance obtained from a log probability plot.

An estimate of the annual runoff can be obtained employing the volume runoff coefficient which is related to percent impervious area as shown in Figure 3-2 and equation (3-5).

$$R_v = R_f C_v \tag{3-5}$$

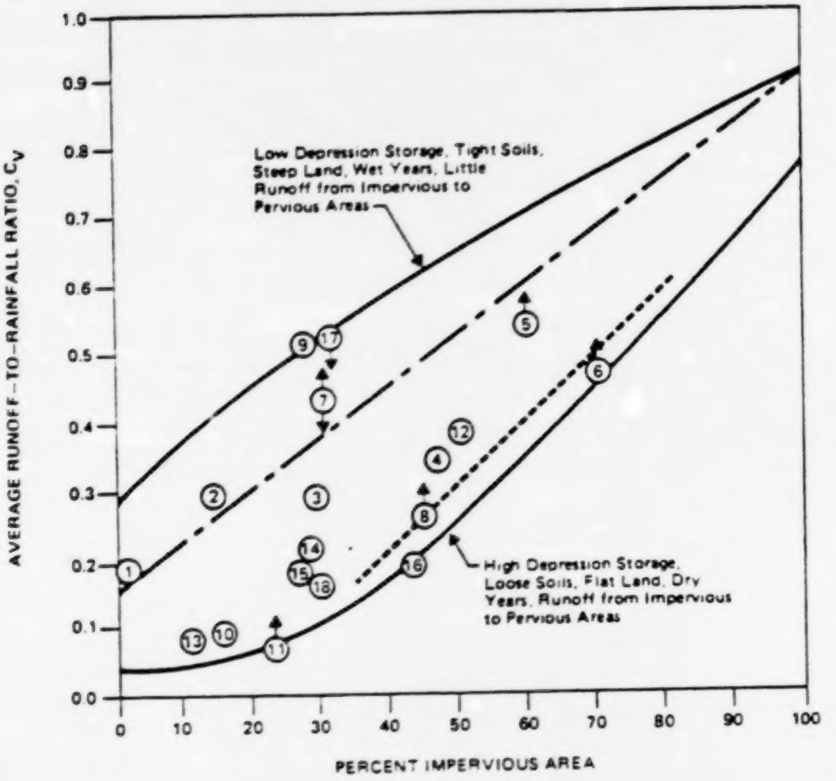
where

R_v = Runoff volume

R_f = Rainfall

C_v = Volume Ratio.

If the latter approach is used, site-specific measurements of concentration and runoff volume ratio may be supplemented by the data from other similar sites available in the literature. If models are employed to generate



STUDY LOCATION CODE

① Upper White Rock, Dallas
② Lower White Rock, Dallas
③ Bachman Branch, Dallas
④ Turtle Creek, Dallas
⑤ Hendrix Creek, New York City
⑥ Spring Creek East, New York City
⑦ Thurston Basin, New York City
⑧ Fourth Creek, Knoxville
⑨ Third Creek, Knoxville
⑩ First Creek, Knoxville
⑪ Plantation Hills, Knoxville
⑫ Northampton, England
⑬ Brighouse, England
⑭ Bradford, England
⑮ Madison, Wisconsin
⑯ Tulsa, Oklahoma
⑰ Durham, North Carolina
⑱ Trout Run, Roanoke, Virginia

LEGEND

—— - —— "Storm" Equation Estimate (12)

▪▪▪▪▪▪▪▪▪▪ Extension of Miller and Viessman (11)

○ Suspect Error, Direction of Probable Correction

Source: EPA (6)

Figure 3-2. Relationship between impervious area and runoff-to-rainfall ratio.

loads, site-specific information on wash-off and build-up rates will usually have to be supplemented by information in the literature.

3.1.3.4 Non-Point Source Loads--

Non-point source loads will have to be estimated based on land use. The analyst has a choice of models which usually employ the Universal Soil Loss equation with yield coefficients or data on areal loadings from various land uses and soil types. The non-point source estimates should be checked against some tributary monitoring data to insure that the magnitude of estimated non-point source loads is realistic. This can be accomplished by sampling one tributary with representative land use over a year and comparison of the estimated non-point source annual load with the measured tributary load less any point sources or urban runoff loads entering the tributary. Experience in the Lake Erie Basin has indicated that, for drainage basins with high clay content soils, tributary sampling during major runoff events is required to obtain an estimate of the total annual load. In extreme cases, 30 percent of the annual phosphorus load from a tributary may enter the lake during a relatively few high tributary flow events associated with storms.

Tributary loads can vary from year to year as a result of the type of water year during which measurements are obtained. The intermittent nature of the transport of nutrients associated with particulates is usually responsible for much of the variation. Erosion varies with the water year. In addition, the particulate component of the load alternately settles and resuspends by scour; thus, this portion of the load has a travel time from source to lake which may be substantially longer than the hydraulic time of travel.

Estimates of nutrients from erosion of banks and lake shoreline should be included in the analysis particularly when total nutrient levels are being considered.

Malfunctioning or improperly installed spetic tanks and other on-lot disposal systems may also represent significant non-point nutrient sources. Estimates of this contribution should be included with non-point source estimates

In studies with limited resources, the areal loading may be used to estimate the contribution from small urban areas.

3.1.3.5 In-Lake Sources--

The primary in-lake source of concern is nutrient release from bottom sediments. This source is generally associated with anoxic hypolimnion conditions for a part of the year. The first step in attempting to evaluate this potential source should involve examination of historical data for low or zero dissolved oxygen levels in the hypolimnion or other sections of the lake. If low dissolved oxygen concentrations (below 1 to 2 mg/l) are suspected, consideration should be given to carrying out a measurement program in the area of concern. Data should be obtained on the seasonal progression of vertical and horizontal structure measuring temperature, dissolved oxygen, limiting nutrients, etc. The data collected can be analyzed using mass balance formulations which include vertical dispersion and settling to determine if the bottom release of nutrients is significant. In general, projects with low bottom dissolved oxygen conditions should consider employing one of the time varible analyses techniques discussed, rather than the steady-state graphical analysis. The reason for this is that the time variable analysis can yield comparisons of calculated and observed nutrient levels which will provide information on the significance of bottom sources of nutrients. If bottom sources of nutrients are significant for all portions of the times when water quality problems are observed, nutrient controls at external sources may not be effective at all or may have a smaller than anticipated impact. Thus, the very objectives and goals of the waste load allocation would be in jeopardy.

3.2 SIMPLIFIED LAKE NUTRIENT MODELS

Over the past decade considerable effort has been devoted to developing a simplified eutrophication analysis framework which could be used to evaluate the trophic state of a lake under present nutrient discharge conditions and predict a future trophic state under modified future nutrient discharges. The models developed involve two distinct steps: first, establishing a causal relationship between nutrient loadings and lake nutrient concentrations, and second, establishing a basis for assigning the lake a trophic state based on lake nutrient concentration.

Models of lake nutrient concentrations either involve the use of the conservation of mass (mass balance) principle or are direct empirical correlations between pertinent lake characteristics and observed lake concentrations. The former will be discussed first.

3.2.1 Nutrient Mass Balance Model

In the simplified analysis illustrated in the top of Figure 3-1, the following assumptions are made:

- the lake is completely mixed
- the lake is at a steady state condition
- total nutrients (dissolved and particulate) are analyzed
- net sedimentation of nutrients occurs.

The general mass balance equation for any substance in a completely mixed lake subjected to a net removal mechanism whose removal rate is assumed proportional to the lake concentration is:

$$V \frac{dp}{dt} = \sum Q_i P_i - K_s pV - Qp \qquad (3\text{-}6)$$

where

$\sum Q_i P_i$ = the sum of all the mass rates of total nutrients discharged to the lake from all sources (PS & NPS) (M/T), (Q_i = flow, L^3/T; P_i = concentration, M/L^3)

p = lake nutrient concentration (M/L^3)

V = lake volume (L^3)

K_s = net sedimentation rate of nutrient (T^{-1})

Q = lake outflow (L^3/T)

Assuming a steady state, e.g., dp/dt = 0, and letting $W = \sum Q_i P_i$, equation (3-6) becomes:

$$0 = W - (K_s V + Q)p$$

$$\text{or } p = \frac{W}{Q + K_s V} \qquad (3\text{-}7)$$

Noting that $V = A\bar{z}$ (A = surface area, $\bar{z}$ = mean depth), Equation (3-7) can be rearranged as:

$$p = \frac{W/A}{(Q/V)\bar{z} + K_s\bar{z}}$$

Letting $W/A = W'$ where $W' =$ areal loading rate $(M/L^2 - T)$

$Q/V = \rho$ where $\rho = 1/\tau\ (T^{-1})$

$\tau = V/Q$ where $\tau =$ hydraulic detention time (T)

$$p = \frac{W'}{\bar{z}(\rho + K_s)} \qquad (3\text{-}8)$$

Typical units employed are: $W'(gm/m^2 - yr)$, $z(m)$, ρ and K_s (yr^{-1}) and $p(gm/m^3 = mg/l)$.

3.2.2 Use of Phosphorus as the Limiting Nutrient

Equation (3-8) is recognized as the form used by Vollenweider in relating phosphorus, nitrogen and other parameters to the lake areal loading rate (28, 29, 30, 31). Since then, work in the field has been concentrated on using total phosphorus, rather than nitrogen, as the single nutrient to describe trophic state and control eutrophication. Reckhow (32) notes that phosphorus was selected since it is generally considered the most manageable of the nutrients and he further cites Sawyer's (33) reasons for the selection:

1. existence of a proven technology for removal of phosphorus from wastewaters
2. significant portions of phosphorus in domestic wastewaters, and all phosphorus in some industrial wastes, are contributed by controllable synthetic detergents
3. phosphorus limitation seems to be the only known means to control the growth of nitrogen-fixing blue-green algae.

Recent work reported by Rast and Lee (34) on 33 lakes and impoundments in the United States, indicated that most of the water bodies were phosphorus limited, primarily on the basis of algal assay procedures. Comparisons of nitrogen:phosphorus ratios for a range of algal species with the ratios of observed inorganic nitrogen:dissolved phosphorus concnetrations in lakes led to similar conclusions as to the lake limiting nutrient.

On the basis of the above reasons, the remainder of the discussion will be focused on the use of total phosphorus in the eutrophication analysis. For lakes, determined to be nitrogen limited, a preliminary methodology and example problem is described in Section 3.2.10.

3.2.3 Total Phosphorus Sedimentation Rate

Equation (3-8) can be used to predict lake phosphorus concentrations if the net sedimentation rate can be estimated. Vollenweider (31) reported that this loss rate (K_s) could be approximated as:

$$\ln K_s = \ln 5.5 - 0.85 \ln \bar{z} \qquad (r = 0.79)$$

or more approximately by

$$K_s = 10/\bar{z} \tag{3-9}$$

where $K_s(\mathrm{yr}^{-1})$, $\bar{z}(\mathrm{m})$.

As noted by both Thomann (35) and Reckhow (32), a constant net sedimentation velocity (v_s) of 10 m/yr is implied in Equation (3-9), where $K_s = v_s/\bar{z}$.

Substitution of this empirical relationship into Equation (3-8) results in:

$$P = \frac{W'}{\bar{z}\rho + v_s} \qquad (3\text{-}10)$$

Based on 14 Canadian lakes, Chapra estimated an apparent settling velocity, v_s, of 16 m/yr (36). Using the same data base, Dillon & Kirchner (39) estimated a value of 13.2 m/yr, which was later refined by Dillon to 12.4 m/yr using a larger data base than used for the preceding estimate (35, 36). In a subsequent publication, Vollenweider (31) revised his empirically based estimate of K_s and deduced a value of:

$$K_s = \rho^{0.5} \qquad (3\text{-}11)$$

Vollenweider (31) cautions that v_s, as used above, is not a real settling velocity but rather is an integration of both positive and negative settling velocities as well as effects due to demineralization, so that using the higher values of real settling velocities would be misleading. Thomann, in a personal communication, suggested that real settling velocities might be able to be used for the total phosphorus in many cases as long as the real settling velocities of the particulate phosphorus forms were adjusted downward to reflect nonsettling of the dissolved phosphorus.

3.2.4 Alternate Form of Mass Balance Equation

Dillon & Rigler (38) calculated the mean annual concentration of total phosphorus, using the mass balance principle and defining a lake retention coefficient, R, as the amount of total phosphorus discharged to the lake which is retained in the lake sediments. Thus, from the basic mass balance, for a steady state,

$$0 = \sum Q_i p_i - R \sum Q_i p_i - Qp$$

or with $\sum Q_i p_i = W$ as previously defined,

$$Qp = W - RW = W(1-R)$$

$$p = \frac{W}{Q}(1-R) = \frac{W'}{Q/A}(1-R)$$

$$p = \frac{W'(1-R)}{q_s} \qquad (3\text{-}12)$$

Dillon used an empirical relationship for R in terms of the water surface overflow rate, $q_s = Q/A$

$$R = 0.426\exp(-0.271\, q_s) + 0.574\exp(0.00949 q_s), \ (r = 0.94).$$

Larsen and Mercier (39), using a data base which included 20 lakes, found that the following empirical relationship fit the data well

$$R = \frac{1}{1 + \rho^{0.5}} \qquad (3\text{-}13)$$

Substitution of (3-11) into (3-8) yields

$$p = \frac{W'}{\bar{z}(\rho + \rho^{0.5})} \qquad (3\text{-}14)$$

and substitution of (3-13) into (3-12) results in

$$p = \frac{W'}{q_s}\left(1 - \frac{1}{1 + \rho^{0.5}}\right)$$

Rearranging and noting that $q_s = Q/A = \bar{z}Q/V = \bar{z}\rho$,

$$p = \frac{W'}{\bar{z}\rho}\,\frac{0.5}{1 + \rho^{0.5}} = \frac{W'}{\bar{z}(\rho + \rho^{0.5})} \qquad (3\text{-}15)$$

is identical to Equation (3-14). Thus, using the Larsen and Mercier estimate for the retention coefficient and the latest Vollenweider loss coefficient reduces the two mass balance models to one and the same equation.

3.2.5 Comparison of Steady-State Mass Balance Equations

Two equations to estimate the total phosphorus concentration are then available, one using a net loss coefficient of $K_s = v_s/\bar{z}$ (Equation 3-10) and the other using $K_s = \rho^{0.5}$ (Equation 3-15).

A comparison of the predictions of the two equations was made. On a unit areal loading basis then, with $v_s = 10$ m/day,

$$p/W' = [\bar{z}\rho + v_s]^{-1} \qquad \text{from Equation (3-10)}$$

will be compared with

$$p/W' = [\bar{z}(\rho + \rho^{0.5})]^{-1} \qquad \text{from Equation (3-15).}$$

The comparison was performed for reasonable bounds on depths ($\bar{z}$) for various hydraulic detention times (τ). As indicated in Table 3-1, the agreement is good for a fairly large range in values of $\bar{z}$ and τ. Some differences are observed for lakes with detention times at and above ten years.

Since reasonable agreement can exist for a reasonable number of lakes and a physical connotation can be retained for the net sedimentation rate, it is recommended that Equation (3-10) be used as the basic predictive model.

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Table 3.1. VALUES OF P/W' for $K_s = V_s/\bar{z}$ and $K_s = \sqrt{\rho}$ USING EQUATIONS 3-10 AND 3-15

$\bar{z}$ (m)	K_s[2]	τ (yr)				
		0.01	0.1	1.0	10	100
1	(a)	0.0091	0.050			
	(b)	0.0091	0.076			
2	(a)	0.0048	0.033			
	(b)	0.0045	0.038			
5	(a)	0.0020	0.017	0.067		
	(b)	0.0018	0.015	0.100		
10	(a)		0.0091	0.050	0.091	
	(b)		0.0076	0.050	0.120	
20	(a)			0.033	0.083	
	(b)			0.025	0.120	
50	(a)				0.067	0.095
	(b)				0.048	0.182
100	(a)					0.091
	(b)					0.091
200	(a)					0.083
	(b)					0.045

[1] $P/W' = [\bar{z}(\rho + K_s)]^{-1}$

[2] (a) $K_s = V_s/\bar{z}$, (b) $K_s = \sqrt{\frac{1}{\tau}}$

3.2.6 Other Nutrient Formulations

In contrast to the preceding mass balance models which estimated coefficients using empirical relationships, a number of investigators (28, 40, 32) have directly correlated lake total phosphorus concentrations to pertinent lake characteristics including areal loading, volumetric loading depth, hydraulic residence time, and surface overflow rate.

As an example, for lakes with $\bar{z}/\tau < 50$ m/yr, Reckhow (32) proposes:

$$p = \frac{W'}{\frac{18\bar{z}}{10+\bar{z}} + 1.05\,\frac{\bar{z}}{\tau}\,\exp\left(0.012\,\frac{\bar{z}}{\tau}\right)}, \quad (r^2 = 0.876)$$

based on data from 33 north temperate lakes. Also, as reported in Reckhow (32), Walker based a relationship on 105 north temperate lakes resulting in:

$$p = \frac{W'\tau}{\bar{z}}\;\frac{1}{1 + 0.824\,\tau^{0.454}}, \quad (r^2 = 0.906)$$

As demonstrated by Reckhow for an example lake situation (Lake Charlevoix), the various mass balance and empirical models yield approximately the same result, although only the empirical methods above were able to generate uncertainty ranges about the predictions.

3.2.7 Determination of Allowable Phosphorus Discharges

The following procedures use trophic states as the basis for determining allowable phosphorus discharges. The procedures are equally applicable to situations where a change in trophic state is not possible but a significant water quality improvement toward a specified target condition can still be achieved. In both cases, the final selection of a control alternative will depend upon the level of improvement and the cost of the proposed processes.

3.2.7.1 Measures of Trophic State--

As indicated in the introduction, four measures of trophic state were cited as guides often used in classifying lakes as oligotrophic, mesotrophic and eutrophic. These were total phosphorus, chlorophyll a, secchi depth, and hypolimnetic oxygen. Primary emphasis in the last decade has been given to total phosphorus and chlorophyll a and the values cited in the introduction are:

Trophic State	Total Phosphorus (μg/l)	Chlorophyll a (μg/l)
Oligotrophic	<10	<4
Mesotrophic	10 - 20	4 - 10
Eutrophic	>20	>10

They appear to provide reasonable bounds in classifying lakes in the north temperate zone into their appropriate trophic states. In some southern and southwestern lakes, higher chlorophyll a and total phosphorus concentrations are sometimes considered acceptable. As discussed in Section 2.1, higher concentrations may also be acceptable in the presence of certain site-specific conditions. In summary, judgment is required in establishing target levels of total phosphorus, chlorophyll a or other measures of lake water quality.

3.2.7.2 Total Phosphorus--

For total phosphorus, Vollenweider (30) compared the lake trophic bounds of 10 to 20 μg/l to investigator-determined trophic status for a number of

European and North American lakes and good agreement resulted as shown in Figure 3-3. It may be noted that slightly higher values of 15 and 30 μg/l were also delineated by Vollenweider, implicitly suggesting that those bounds may be appropriate also. Similar results are shown in the bottom of Figure 3-3 for 33 lakes and impoundments in the United States, as reported by Rast and Lee (34).

3.2.7.3 Chlorophyll a—

As reported in Thomann (35), values of chlorophyll a from Bartsch and Gakstatter (Figure 3-4) indicate that the suggested bounding values of 4 μg/l and 10 μg/l are appropriate for describing the eutrophic state of a lake. Using primarily north temperate lakes, the following relationships between total phosphorus and chlorophyll a concentrations have been suggested (41, 34, 42):

Bartsch and Gakstatter (41)

$$\log_{10} (\text{Chl } \underline{a}) = 0.807 \log_{10} (p) - 0.194 \qquad (3\text{-}16)$$

Rast and Lee (34)

$$\log_{10} (\text{Chl } \underline{a}) = 0.76 \log_{10} (p) - 0.259 \qquad (3\text{-}17)$$

Dillon and Rigler (38)

$$\log_{10} (\text{Chl } \underline{a}) = 1.449 \log_{10} (p_s) - 1.136 \qquad (3\text{-}18)$$

where P and Chl a are the total phosphorus and chlorophyll a concentrations, respectively, in μg/l. In Dillon and Rigler's formula, p_s is the spring total phosphorus value and chlorophyll a is the summer value.

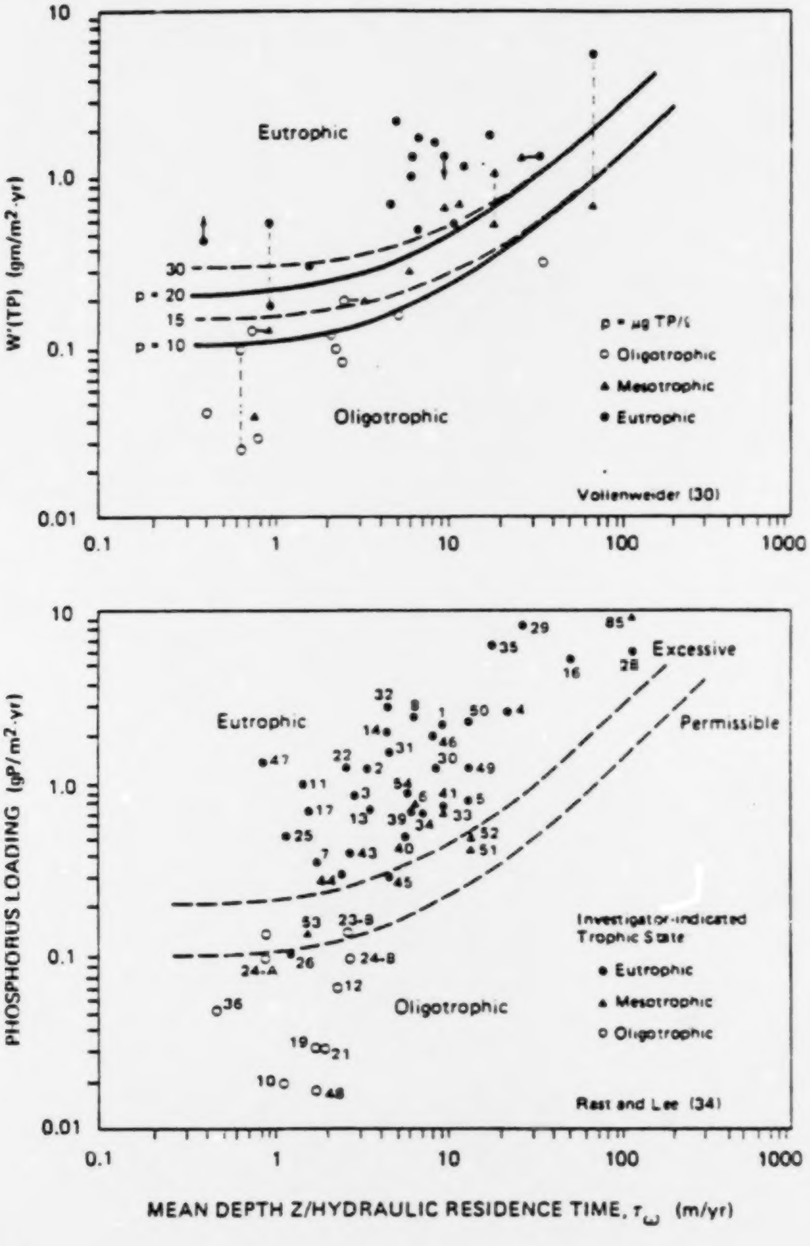
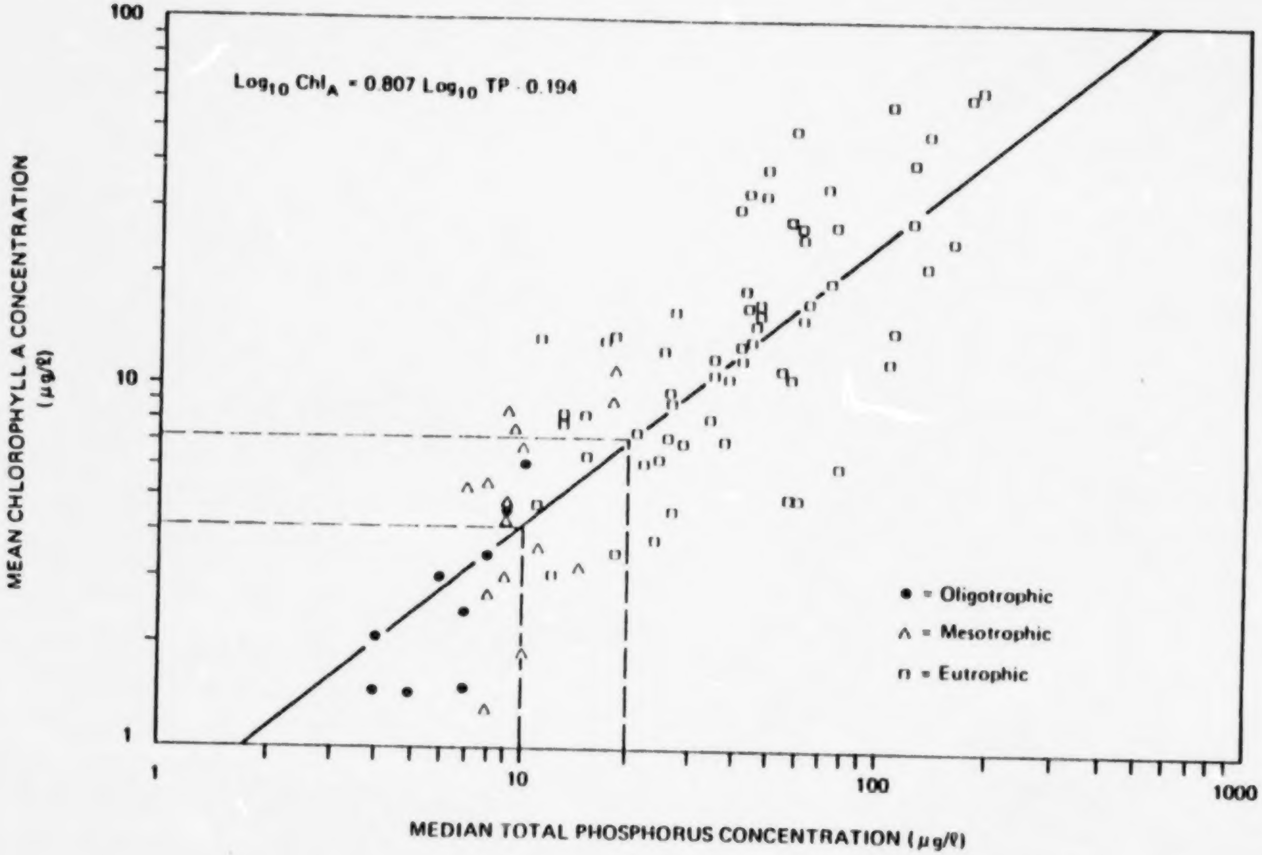


Figure 3-3. Test of trophic state indicators.



Source: Bartsch and Gakstatter (41)

Figure 3-4. The relationship between chlorophyll A and total phosphorus concentrations in northeastern U. S. lakes and reservoirs.

These three formulas are useful in estimating the chlorophyll a concentration resulting from different phosphorus concentrations. This will further illustrate the water quality benefits resulting from various phosphorus control alternatives.

3.2.7.4 Secchi Depth and Dissolved Oxygen--

Rast and Lee (34) report a correlation between secchi depth (z) and total phosphorus (p) of:

$$\log_{10}(z) = -0.359 \log_{10}(p) + 0.925 \tag{3-19}$$

where secchi depth is in meters and total phosphorus is in μg/l for the 33 lakes, and impoundments in the United States. In addition, using data from 13 of the above lakes, plus 21 additional U.S. and Canadian lakes, Rast and Lee report a hypolimnetic oxygen demand of:

$$\log_{10}(S_b) = 0.467 \log_{10}(p) - 1.07 \tag{3-20}$$

where S_b is the areal benthic oxygen demand (gm/m^2 day) and p is the total phosphorus in μg/l. Although to date this predicted benthal oxygen demand has not been significantly used to classify lakes, Reckhow (32) reports that work is underway in this area.

3.2.7.5 Calculation of Allowable Phosphorous Loading--

Allowable loadings may be selected if:

- a trophic state or acceptable water quality condition is specified
- the bounds between trophic states are (or the acceptable condition is) specified in terms of total phosphorus or chlorophyll a.

It is presumed the least eutrophic state consistent with resource constraints will be selected. The boundary concentration between oligotrophic, mesotrophic, and eutrophic must then be selected for either total phosphorus or chlorophyll a.

3.2.7.6 Use of Total Phosphorus to Describe Trophic State--

Values of 10 μg/l and 20 μg/l for total phosphorus are commonly used to distinguish the three trophic states, although some variation in these values is clearly possible based on the data shown in Figure 3-3. It is to be noted that these values are primarily from north temperate lakes and more tropical lakes are clearly outside the data base. It is recommended that local data bases of total phosphorus be reviewed and correlated with perceived trophic states to aid in the selection of trophic boundary concentrations. For data poor systems, use of the 10 and 20 μg/l modified for local conditions can be considered. In addition, some sensitivity analysis could be performed using slightly higher values, and/or 15 and 30 μg/l total phosphorus.

Assuming that 10 and 20 μg/l are selected, Equation (3-10) is then rearranged to yield a predictive equation for the surface areal loading rate as follows:

$$W' = p_c(\bar{z}\rho + v_s)$$

where p_c is the critical boundary total phosphorus concentration in gm/m^3 or mg/l.

$$W'_1 = 0.010\ (\bar{z}\rho + v_s) \qquad (3\text{-}21)$$

$$W'_2 = 0.020\ (\bar{z}\rho + v_s) \qquad (3\text{-}22)$$

where W'_1 and W'_2 are the areal surface loading rates that divide oligotrophic from mesotrophic and mesotrophic from eutrophic conditions, respectively. Plots of equations (3-21) and (3-22) with an assumed value of 10 m/yr are shown in Figure 3-3.

Selection of the appropriate net sedimentation velocity (v_s) should be based on a local data base, with a value of 12.4 m/yr suggested if no data are available; a possible range of (v_s) from 10 to 16 m/yr has been reported.

3.2.7.7 Use of Chlorophyll a to Describe Trophic State--

Chapra and Tarapchak (43) have suggested a procedure for specifying total phosphorus areal loadings in terms of chlorophyll a boundary values between trophic states. Thomann (35) has summarized the procedure as:

1) determine mean annual concentration of total phosphorus
2) estimate the concentration of total phosphorus in the spring using the mean annual concentration
3) compute mean summer chlorophyll a concentrations from spring concentrations of total phosphorus.

The mean annual concentration is calculated from Equation (3-10):

$$p = \frac{W'}{\bar{z}\rho + v_s}$$

The spring total phosphorus (p_s) is calculated from the mean value using a best fit line developed by Chapra and Tarapchak:

$$p_s = 0.9\ p \qquad (3\text{-}23)$$

The summer chlorophyll a concentration is estimated from the Dillon and Rigler correlation of p_s and chl a summer, Equation (3-18):

$$\log_{10}(\text{chl } \underline{a}) = 1.449 \log_{10}(p_s) - 1.136$$

$$\text{or,} \quad \text{chl } \underline{a} = 0.0731\ (p_s)^{1.449} \qquad (3\text{-}24)$$

where chl a and p_s are in μg/l. The allowable phosphorus areal loadings are arrived at by combining equations (3-10), (3-23), and (3-24) to give:

$$W' = 0.0055\ (\text{chl } \underline{a})^{0.69}\ (\bar{z}\rho + v_s) \qquad (3\text{-}25)$$

where W' is in gm/m^2-yr, chl a is in μg/l, $\bar{z}\rho$ and v_s are in m/yr. Chapra and Tarapchak selected 2.75 μg/l and 8.7 μg/l as the boundary chlorophyll concentrations between the three trophic states, resulting in:

$$W'_1 = 0.011\ (\bar{z}\rho + v_s) \qquad (3\text{-}26)$$

$$W'_2 = 0.025\ (\bar{z}\rho + v_s) \qquad (3\text{-}27)$$

Use of values of 4 and 10 μg/l would result in

$$W'_1 = 0.014\ (\bar{z}\rho + v_s) \qquad (3\text{-}28)$$

$$W'_2 = 0.026\ (\bar{z}\rho + v_s) \qquad (3\text{-}29)$$

where W'_1 and W'_2 are the total phosphorus surface areal loading rates for the oligotrophic-mesotrophic and mesotrophic-eutrophic boundaries.

It may be noted that equations (3-26) and (3-21) yield an almost identical value for W'_1, whereas equation (3-27) is somewhat more lenient than equation (3-22) in estimating the value of W'_2.

There are very significant differences in what constitute acceptable chlorophyll a levels. Acceptable levels will vary regionally and may also vary among lakes in the same geographic region due to natural turbidity, depth, and historical water usage. The generalization of the above calculation procedure with assignment of locally applicable values of chlorophyll a and/or total phosphorus as target objective should be considered in waste load allocations.

3.2.8 Calculation Procedure

Based on the foregoing a simplified calculation procedure for nutrient allocations to control lake eutrophication is:

Step 1. Estimate the lake volume, surface area, and mean depth using available bathymetric charts and/or survey data obtained by state and local agencies or academic institutions. The U.S. Geological Survey's (USGS) seven and one-half minute quadrangles may be sufficiently accurate for the larger lakes to estimate surface areas.

Step 2. Estimate the mean annual outflow rate. Ideally, this data would be obtained from a gaging station at the lake outlet. If unavailable, data from a nearby upstream or downstream gage could be used by correcting the flow due to runoff from the drainage area between the gage and the lake outlet. Lacking nearby gaging stations, the drainage area upstream of the lake outlet may be obtained from the seven and one-half minute quadrangle sheets. The product of this area and a flow per unit area (typical of the type of drainage basin) yields the lake outflow. Values of annual and low flow per unit area are available (7).

For lakes with detention times less than the year scale, mean outflow rates should be obtained from a correspondingly shorter flow data base. Where urban areas draining to the lake constitute a significant fraction of the total drainage area, flow estimates from urban runoff and combined sewer overflows should be included in the hydrologic balance around the lake (6). For lakes with large surface areas, surface precipitation and evaporation should also be included.

Step 3. Determine average annual total phosphorus loading due to all sources. These include all tributary inflows, municipal and industrial sources, distributed urban and rural runoff, and atmospheric inputs. Estimation of these loadings is discussed in Section 3.1.3. Lacking an extensive local data base, the methodologies and summary loading tables in (6, 7, 8) should prove useful in making a first estimation.

Step 4. Assign a net sedimentation (loss) rate for total phosphorus consistent with a local data base. (v_s = 12.4 m/yr. if no data are available)

Step 5. Select trophic state objectives of either total phosphorus or chlorophyll a consistent with local experience. Lacking this, total phosphorus limits of 10 μg/l and 20 μg/l to characterize the "permissible" and "dangerous" concentrations can be considered. Calculate values of W'_1 and W'_2 for the specific lake depth and detention time.

Step 6. Compare the total areal loading determined in Step 3 to the values of W'_1 and W'_2 calculated in Step 5. If the lake loading places it in an undesired trophic state, determine the reduction required to return the lake to the desired level.

Step 7. If a reduction is required, determine whether feasible point source controls will accomplish the reduction and allocate among the various point sources.

Step 8. Test the results of the analysis by: selecting higher total phosphorus concentration for the trophic boundaries; using chlorophyll a as the determinant of the trophic boundaries; selecting higher and lower values of the net sedimentation rate; etc.

In the foregoing, a loading plot similar to Figure 3-3 will prove useful in the analysis, especially when curves for W'_1 and W'_2 are drawn on the graph for the various sensitivity analyses of Step 8.

The following example problem illustrates the calculation procedure described above:

Example Problem

Data: Lake Geometry

Volume	V	$= 10 \times 10^6\ m^3$
Surface Area	A	$= 2 \times 10^6\ m^2$
Depth	$\bar{z}$	$= 5\ m$
Outflow Rate	Q	$= 0.3\ m^3/sec$

$$Q = 0.3\ m^3/sec \times 3.154 \times 10^7\ sec/yr = 9.46 \times 10^6\ m^3/yr$$

Discharges of Total Phosphorus

Point Sources	= 400 kg/yr
Non-Point Sources	= 500 kg/yr

Problem: Determine required reductions in the point or non-point source discharge to classify the lake as marginally mesotrophic and oligotrophic.

Solution:

1. Select a net sedimentation rate of 12.4 m/yr.
2. Assume total phosphorus trophic boundaries of 10 $\mu g/l$ and 20 $\mu g/l$.

$$W'_1 = 0.010\ (\bar{z}\rho + 12.4) \qquad (3\text{-}21)$$

$$W'_2 = 0.020\ (\bar{z}\rho + 12.4) \qquad (3\text{-}22)$$

3. With $\tau = V/Q = 10 \times 10^6\ m^3/9.46 \times 10^6\ m^3/yr = 1.06\ yr$,

$\rho = 1/1.06 = 0.943\ yr^{-1}$, and

$\bar{z}\rho = 5\ m \times 0.943\ yr^{-1} = 4.7\ m/yr$,

$$W'_1 = 0.010\ (4.7 + 12.4) = 0.171\ \text{gm/m}^2\text{-yr}$$

$$W'_2 = 0.020\ (4.7 + 12.4) = 0.342\ \text{gm/m}^2\text{-yr}$$

4. The total lake areal loading is:

$$W' = \frac{(400 + 500)\ \text{kg/yr} \times 1000\ \text{gm/kg}}{2 \times 10^6\ \text{m}^2}$$

$$= 0.451\ \text{gm/m}^2\text{-yr}$$

5. To become marginally mesotrophic, a reduction of

$$0.451 - 0.342 = 0.109\ \text{gm/m}^2\text{-yr}$$

is required, or

$$0.109\ \text{gm/m}^2\text{-yr.} \times \frac{2 \times 10^6\ \text{m}^2}{10^3\ \text{gm/kg}} = 218\ \text{kg/yr}$$

This requires a reduction in the point sources of

$$(218/400) \times 100 = 55\%$$

6. To become marginally oligotrophic, a reduction of

$$0.451 - 0.171 = 0.280\ \text{gm/m}^2\text{-yr}$$

or $0.280 \times 2 \times 10^6/10^3 = 560$ kg/yr is required.

Since the point sources only amount to 400 kg/yr, the desired trophic status could not be attained by only point source control.

7. A loading plot with the selected W'_1 and W'_2 curves is shown in Figure 3-5, together with the location of the lake on the plot for three loading conditions: present, 55 percent point source load reduction and 100 percent point source removal.

3.2.9 Comments on Limitations and Applicability

1. Reckhow (32) advises caution in applying the methods herein to shallow lakes (depths less than approximately three meters), since he has often found unpredictable behavior in the total phosphorus con-

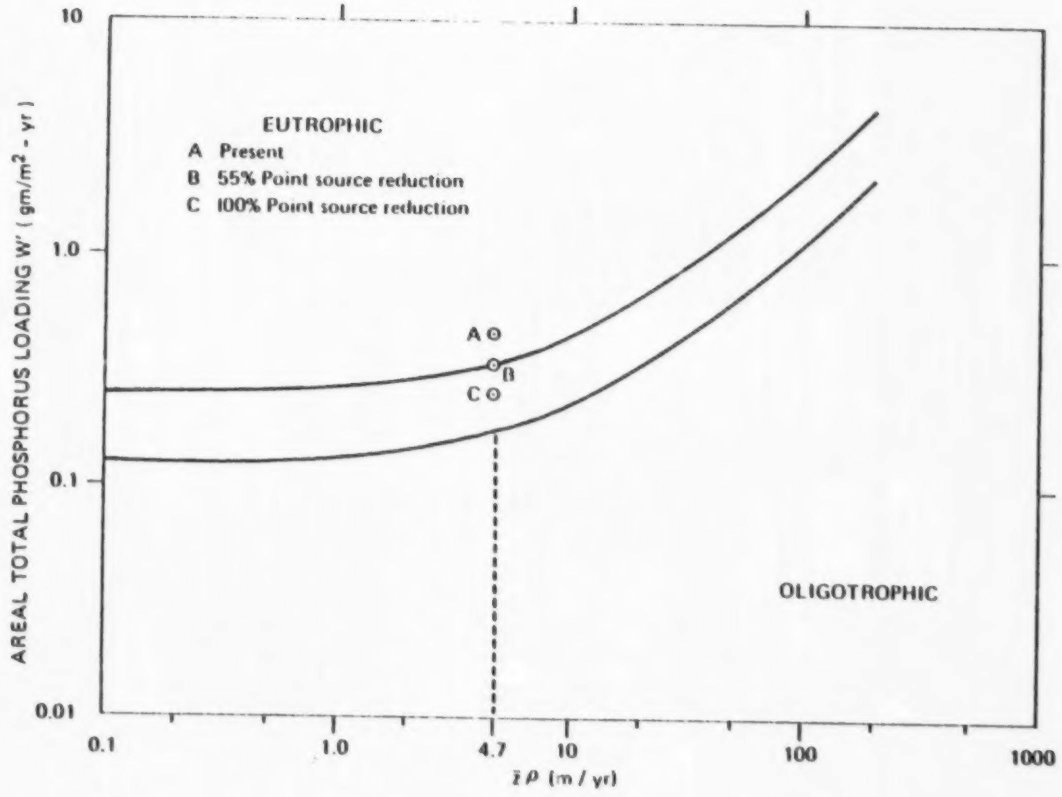


Figure 3.5. Effect of point source control on trophic status, sample problem.

centrations. He suggests that the potential for mixing of the sediments (wind induced) may be a factor. Of over 75 lakes included in the data base (29, 31, 34, 39, 44), less than ten percent had depths less than three meters (see Table 3.2)

Rast and Lee (34) note that the Vollenweider approach may not be applicable to impoundments with hydraulic detention times in order of a month or less, especially for those with marked stratification of inflowing waters during the growing season. In addition, they observe that the critical loading criteria may have to be modified for lakes with excessive macrophytes and attached algae, because the criteria were developed for planktonic algae.

2. Chapra and Tarapchak (43) note that the assumption of steady state is reasonable when, on an annual basis, the morphometry, climate and nutrient supply are constant year to year. In the case of lakes undergoing severe cultural eutrophication, an accumulation term must be added to the mass balance equation to properly characterize lake concentrations. If not accounted for, unrealistic sedimentation coefficients might be selected which would lead to unconservative predictions.

3. Dillon (October 1971) discussed lake restoration projects and reported that for the Zellersee (Switzerland) and Lake Washington (Washington) marked improvements have been noted after significant phosphorus reductions. However, in Lakes Sammamich (Washington) and Norrviken (Sweden), in spite of significant phosphorus reductions--the areal loads remained in excess of permissible levels -- no improvements have been noted. Dillon postulates that wind-generated mixing may be regenerating sediment phosphorus in L. Norrviken, a shallow lake ($\bar{z} = 5.4$ m). In Lake Monona (Wisconsin), after removing a point source, copper sulfate is needed twenty years after the diversion to control algae. However, "high loading" of approximately 2 gm/m^2-yr is still present due to agricultural drainage.

   Little Otter Lake (in Ontario, Canada) ($\bar{z} = 2.7$ m, $\tau = 0.1$ yr), which had severe eutrophication problems due to a single industrial point source of poly-phosphate, recovered rapidly upon removal of the discharge. On the other hand, the Rotsee (Switzerland), of small size and with agricultural drainage, showed no improvement when a wastewater diversion project was completed. Finally, Denmark's Lyngby-So, after a diversion of sewage, improved for four years then incurred a significant macrophytic growth. Dillon theorizes that this may have been due to increased light penetration occasioned by decreased phytoplankton concentrations.

4. The data base upon which the analysis framework has been tested is almost exclusively from north temperate lakes. Although the basic mass balance model may still yield good results for more tropical

Table 3.2. CHARACTERISTICS OF SELECTED LAKES IN SIMPLIFIED EUTROPHICATION ANALYSIS DATA BASE

No.	Name	Depth (m)	τ_o (yr)	Areal Loading (gm/m^2-yr) P	N	Tropic State	Ref.
Switzerland							
1	Agerisee	48	8.70	0.16		O	29,44
2	Baldeggersee	34	4.55	1.75		E	29,44
3	Bodensee-Obersee	100	4.88	4.07		M	29,44
4	Greifensee	19	2.04	1.57		E	29,44
5	Hallwilersee	28	3.85	0.55		E	29,44
6	Lac Leman	154	12.00	0.79		M	44
7	Pfaffikersee	18	2.60	1.36		E	29,44
8	Turlersee	14	2.15	0.30		M	29,44
9	Zellersee	37	2.70	1.20		E	44
Sweden							
10	Hjalmaren	6	3.6	0.30		E	44
11	Malaren	12.5	2.7	0.70		E	44
12	Norrviken	5.4	0.571	2.1('70)		E	44
13	Vattern	39	56.0	0.065		O	44
14	Vanern	25	8.3	0.15		O-M	44
Italy							
15	Maggiore	177	4	3		M	31
Canada							
16	Beech	9.8	0.0441	1.68			44
17	Bob	18.0	2.7	0.16			44
18	Cameron	7.1	0.0529	2.21			44
19	Clear	12.5	7.7	0.040		O-M	39,44
20	Cranberry	3.5	0.0159	1.28			44
21	Eagle-Moose	12.8	0.493	0.23			44
22	ELA 227	4.4	4.2	0.34		E	39,44
23	Four Mile	9.3	3.8	0.11			44
24	Green	6.1	0.0260	1.77			44
25	Halls	27.2	1.0	0.22			44
26	Kamalka	58.0	110	0.32		E	39,44
27	Maple	11.6	0.13	0.86			44
28	Oblong-Haliburton	17.7	3.1	0.12			39,44
29	Okanagan	75.3	59	0.39		O	39,44
30	Pine	7.4	0.054	1.06			44
31	Raven	0.73	0.067	0.22			44
32	Skaha	26.5	1.1	2.20		M-E	39,44
33	Talbot	0.85	0.20	0.10			44
34	Twelve Mile-Bashhung	18.1	0.42	0.35			44
35	Wood	21.0	110	0.50		E	39,44

Table 3.2. CHARACTERISTICS OF SELECTED LAKES IN SIMPLIFIED EUTROPHICATION ANALYSIS DATA BASE (concluded)

No.	Name	Depth (m)	τ_o (yr)	Areal Loading (gm/m^2-yr)		Tropic State	Ref.
				P	N		
	United States						
36	Blackhawk WI	4.9	0.5	2.2	23.4	E	34
37	Brownie MN	6.8	2.0	1.18		E	34
38	Calhoun MN	10.6	3.6	0.86		E	34
39	Camelot-Sherwood WI	3	0.09-0.14	2.5	34.6	E	34
40	Canadaroga NY	7.7	0.6	0.8	18.0	E	34
41	Cayuga NY	54	8.6	0.8	14.3	M	34
42	Cedar MN	6.1	3.3	0.35		E	34
43	Cox Hollow WI	3.8	0.5-0.7	1.8	19.1	E	34
44	DogFish MN	4.0	3.5	0.02		O	34
45	Dutch Hollow WI	3	1.8	1.0	10.4	E	34
46	Erie	18	2.6	1.06		E	44
47	George NY	18	8	0.07	1.8	O-M	34
48	Harriet MN	8.8	2.4	0.71		E	34
49	Huron	61	21	0.13		O-M	31
50	Isles MN	2.7	0.6	2.03		E	34
51	Kegonsa WI	4.6	0.35	6.64		E	39,44
52	Kerr(Roanoke) NC	10.3	0.2	5.2	36.2	E	34
53	Kerr (Nutbush)VA	8.2	5.1	0.7	2.4	M	34
54	Lamb MN	4.0	2.3	0.03		O	34
55	Meander MN	5.0	2.7	0.03		O	34
56	Mendota WI	12	4.5	1.2	13	E	34
57	Menona WI	7.8	1.2	2.14		E	44
58	Michigan-open waters	84	30-100	0.10	1.3	O	34
59	Minnetonka MN	8.3	6.3	0.1-0.2('73)		E	34
60	Ontario	84	7.9	0.65-0.86		M	31
61	Redstone WI	4.3	0.7-1.0	1.5	18	E	34
62	Sallie MN	6.4	1.1-1.8	1.5-4.2	2.8-3.0	E	34
63	Sammamish WA	18	1.8	0.7	13.0	M	34
64	Sebasticook ME	6.0	2.6	0.21		E	44
65	Shagawa MN	5.7	0.8	0.7	7.8	E	34
66	Stewart WI	1.9	0.08	4.8-8.0	73.6	E	34
67	Superior	148	185	0.03		O	31
68	Tahoe NV	313	700	0.05	0.52	O	34
69	East Twin OH	5.0	0.5-0.9	0.5-0.8	19-31	E	34
70	West Twin OH	4.34	1.0-1.8	0.2-0.4	15-16	E	34
71	Twin Valley WI	3.8	0.4-0.5	1.7-2.0	17	E	34
72	Virginia WI	1.7	0.9-2.8	1.2-1.5	18.3	E	34
73	Waldo OR	36	21	0.017	0.33	O	34
74	Washington WA	33	2.4	1.2-2.3	8-19	E('69)	34
				0.47	4.4	M('74)	34
75	Waubesa WI	4.8	0.30	9.93		E	39,44
76	Weir FL	6.3	4.2	0.14	2.6	M	34
77	Wingra WI	2.4	0.4	0.9	5.1	E	34

O = Oligotrophic
M = Mesotrophic
E = Eutrophic

lakes, it is possible that the net sedimentation rate may be different and caution must be used. In addition, acceptable levels of total phosphorus and chlorophyll a may be substantially higher in southern lakes or those with inputs of high clay content soils.

3.2.10 Preliminary Nitrogen Allocation

As indicated previously, most recent investigations of the simplified eutrophication method have been restricted to the analysis of total phosphorus and there is no equivalent data base to support a similiar methodology for nitrogen. For those cases where nitrogen has been identified as the limiting nutrient, a tentative procedure -- initially suggested by Vollenweider (28) and discussed by Rast and Lee (34) -- might be to utilize the total phosphorus methodology for total nitrogen, after suitably adjusting the trophic boundary loading curves.

This procedure is recommended only as an interim measure until further investigations (Rast and Lee) produce trophic boundary concentrations and appropriate removal coefficients for total nitrogen. It should be recognized that the use of nitrogen:phosphorus ratios is imprecise, and any interpretation of data based on these ratios should recognize the potential for error. This type of analysis may be sufficient as a screening tool for nitrogen control but may not be adequate to justify the need for expensive nitrogen treatment.

Assuming for algae, a stoichiometric nitrogen to phosphorus ratio of 7.2:1 (34), the total nitrogen loads at the trophic boundaries would be 7.2 times those for total phosphorus. Vollenweider (28) found these values too conservative and suggested using a ratio of 15:1. Using data from the National Eutrophication Survey and six Swiss lakes (34, 44), the 15:1 ratio

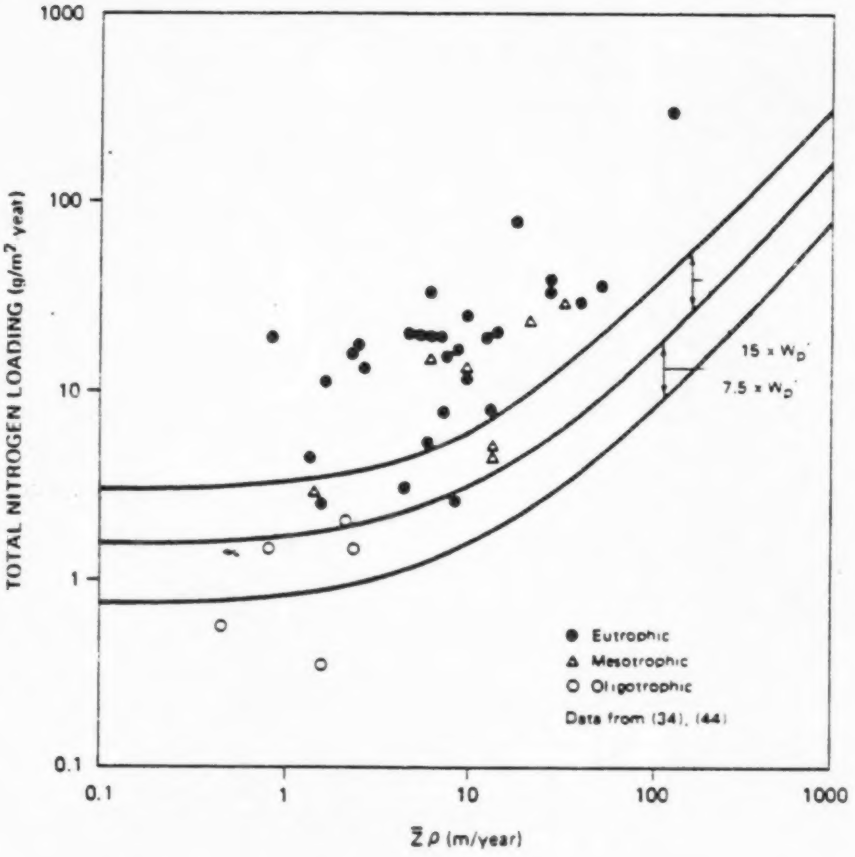


Figure 3-6. Total nitrogen loading plot.

appears to fit the data best, as seen in Figure 3-6. Thus, preliminary trophic boundary areal loads are:

$$W'_{1_N} = 0.15(z + v_{s_N}) \tag{3-30}$$

$$W'_{2_N} = 0.30(z + v_{s_N}) \tag{3-31}$$

where the subscript N refers to nitrogen. Assuming denitrification and nitrogen fixation are not significant, the net sedimentation rate of total nitrogen may be similar to that for total phosphorus and values of V_{s_N} could be set equal to 10 to 16 m/yr. The following example problem illustrates the calculation procedure described above:

Example Problem

Data:

Lake Geometry: same as in example problem of Section 3.2.8

($\bar{z}$ = 4.7 m/yr)

Outflow rate: same as in previous example problem

($A = 2 \times 10^6 \ m^2$).

Discharges of Total Nitrogen

Point Sources = 8000 kg/yr
Non-Point Sources = 4500 kg/yr

Problem: Estimate the required reduction in the point or non-point source load to classify the lake as marginally mesotrophic.

Solution:

1. Assume a net sedimentation rate of total nitrogen of 10 m/yr.
2. Assume trophic boundaries for total nitrogen fifteen times those of total phosphorus, that is, 150 and 300 μg/l, so that you have:

$$W'_{1_N} = 0.15\ (4.7 + 10) = 2.21\ gm/m^2\text{-}yr$$

$$W'_{2_N} = 0.30\ (4.7 + 10) = 4.41\ gm/m^2\text{-}yr$$

3. The present areal loading is:

$$W' = \frac{(8000 + 4500)\ \text{kg/yr} \times 1000\ \text{gm/kg}}{2 \times 10^6\ \text{m}^2} = 6.25\ \text{gm/m}^2\text{-yr}$$

4. To become marginally mesotrophic, a reduction of

$$6.25 - 4.41 = 1.84\ \text{gm/m}^2\text{-yr}$$

is required, or

$$1.84\ \frac{\text{gm}}{\text{m}^2\text{-yr}} \times \frac{2 \times 10^6\ \text{m}^2}{1000\ \text{gm/kg}} = 3680\ \text{kg/yr}$$

This requires a reduction in the point source of

$$\frac{3680}{8000} \times 100 = 46\%$$

3.3 TIME VARIABLE MASS BALANCE MODELS

Models in this category are extensions of the approach developed by Vollenweider (30). The basic mass balance equations for total phosphorus in a completely mixed lake, which Vollenweider solved for steady state, are employed with provision for flows and loads which vary with time. The resultant formulations calculate concentrations that could represent lake-wide average total phosphorus concentrations which are a function of time. The calculated time history of phosphorus can then be compared to observed phosphorus concentrations to provide calibration for the analysis framework. A lumped parameter is employed to represent losses of phosphorus from the system.

Two interesting site specific applications of time variable mass balance models were developed by Chapra (45) and Larsen (39). The Chapra application considered the Great Lakes eutrophication problem and employed seven completely mixed segments similar in concept to the work of O'Connor (46). The Great Lakes calculation included a historical simulation extending from 1800 to 1970, as well as predictive simulation extending from 1970 to 2000. The time scale of this analysis was decades with a yearly time step. By contrast, the work of Larsen and coworkers on Shagawa Lake considered an annual simulation covering the period 1971 to 1976 with what appeared to be daily to weekly time steps. The two site-specific projects indicate how the analysis framework can be adjusted to the time and space scale of the problem. The Great Lakes time and space scales are large while Shagawa Lake had a detention time ranging from 0.42 to 1.23 years for the period investigated.

3.3.1 Formulations

The phosphorus residence time analysis indicated in Figure 3-7 and Equation (3-22) considers a mass balance for a completely mixed lake.

$$\frac{dP_r}{dt} = \frac{W}{V} - \frac{QP_r}{V} - \frac{S}{V} + \frac{B_s}{V} \qquad (3\text{-}32)$$

where

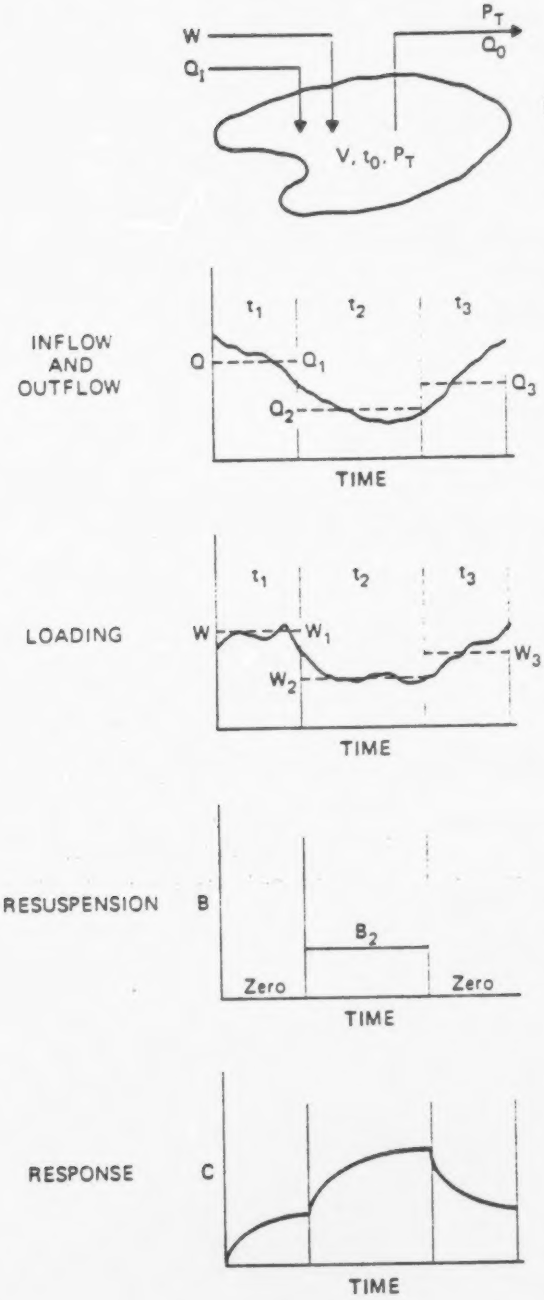
P_r = total phosphorus concentration (M/L^3)
W = mass loading rate of total phosphorus (M/T)
V = lake volume (L^3)
Q = lake outflow (L^3/T)
S = lumped phosphorus removal parameter (M/T)
B_s = lumped internal source of phosphorus parameter (M/T)

There are several methods of solving equation (3-32). One method employs numerical integration usually utilizing a computer. The equation is not complex, and the input data will usually be small; therefore, a mini-computer could be considered. A second method of solving equation (3-32) is through integration with Q, W, V, S, B_s as constants. The resultant solution is presented in equation (3-33) for boundary conditions $t = 0$, $P_r = P_o$ and $t = t$, $P_r = P_T$:

$$P_T = \frac{W-S+B_s}{Q}\left(1 - \exp(-t/t_o)\right) + P_o\exp(-t/t_o) \qquad (3\text{-}33)$$

where

t_o = lake detention time V/Q
P_o = initial phosphorus concentration at $t = 0$
t = time interval for which calculation is to be made.


W
Q_I
P_T
Q_0
V, τ_0, P_T
INFLOW AND OUTFLOW
τ_1
τ_2
τ_3
Q
Q_1
Q_2
Q_3
TIME
LOADING
τ_1
τ_2
τ_3
W
W_1
W_2
W_3
TIME
RESUSPENSION
B
B_2
Zero
Zero
TIME
RESPONSE
C
TIME

$$P_T = \frac{W_n + B_{s_n}}{V(\frac{1}{\tau_0} + \upsilon)} (1 - e^{-t(\frac{1}{\tau_0} + \upsilon)}) + P_0 e^{-t(\frac{1}{\tau_0} + \upsilon)}$$

υ = constant

Time Segment τ_1

$Q = Q_1$
$W = W_1$
B_s = Zero
$P_0 = P_0$

Time Segment τ_2

$Q = Q_2$
$W = W_2$
$B_s = B_2$
$P_0 = P_T$ @ τ_1

Time Segment τ_3

$Q = Q_3$
$W = W_3$
B_s = Zero
$P_0 = P_T$ @ τ_2

Figure 3-7. Phosphorus mass balance for completely mixed lake.

Equation (3-33) can be solved on a calculator or with a computer. The approach as shown in Figure 3-6 and the example is to subdivide the time over which the calculation is to be carried out into time segments, which may have different lengths t_1, t_2, ...t_n, but with constant Q, W, V, S, and B_s in each of the time segments. The calculation should employ the first observed phosphorus concentration for the value of P_o at $t = 0$; P_T is then calculated using equation (3-33) for any times "t" less than the first time segment length t_1. The value of P_T is then calculated at t_1 and this value of P_T is substituted for P_o, and the calculation is repeated for any desired times in the next time segments t_2-t_1. The procedure may be repeated for as many time segments as required. The procedure could even employ a daily time step. Selection of the calculation method is really a matter of preference for the analyst.

The actual formulation employed by Larsen et al. (39) included a lumped first order settling term v rather than a phosphorus removal parameter S. Equation (3-34) is the differential equation and equation (3-35) is the solution for boundary conditions $t = 0$, $P_r = P_o$ and $t = t$, $P_r = P_T$.

$$\frac{dP_r}{dt} = \frac{W+B_s}{V} - \frac{QP_r}{V} - vP_r \qquad (3\text{-}34)$$

$$P_T = \frac{W+B_s}{V(1/t_o+v)}\left(1 - \exp{-t}\,(1/t_o + v)\right) + P_o \exp{-t}(1/t_o + v) \qquad (3\text{-}35)$$

The identical solution techniques available for equations (3-32) and (3-33) can be employed to solve equations (3-34) and (3-35). The following example illustrates the use of the time variable model described above.

3.3.2 Example Problem

Data:

$V = 10^7\ m^3$
$\underline{A} = 2 \times 10^6\ m^2$
$\underline{z} = 10^7/2 \times 10^6 = 5\ m$
$v' = 0.1\ m/day$
$v = 0.1/5 = 0.02/day = 0.14/wk.$
$Q = 0.3\ m^3/sec = 9.46 \times 10^6\ m^3/yr = 1.82 \times 10^5\ m^3/wk$
$t_o = 10^7\ m^3/1.82 \times 10^5\ m^3/wk = 54.95\ wks$

$$1/t_o + v = \frac{1}{54.95} + 0.14 = 0.1582/wk$$

Problem: A lake is subjected to a loading of 17.3 kg/wk for 46 weeks and a bottom loading of 3 $mg/m^2/day$ for the last six weeks (week 40-46). Calculate the phosphorus concentration in the lake at the end of the following weeks: three, fifteen, twenty-five, forty, and forty-six. The initial phosphorus concentration of the lake is zero.

Solution:

Period #1 (From 0 to 40 Wks)

$W = 17.3\ kg/wk$

$B_s = 0$

$P_o = 0$

$$P = \frac{W + B_s}{V\,(1/t_o + v)}\left(1 - \exp{-t(1/t_o + v)}\right) + P_o \exp{-t(1/t_o + v)}$$

t = 3 wks.

$$P = \frac{17.3 \text{ kg/wk} + 0}{10^7 \text{ m}^3 \text{ (0.1582/wk)}}\left(1 - \exp{-3} \times 0.1582\right) + 0$$

$$= 1.0936 \times 10^{-5} \frac{\text{kg}}{\text{m}^3} \quad (1 - .622) = 0.413 \times 10^{-5} \text{kg/m}^3 = 0.004 \text{ mg/l}$$

t = 15 wks

$$P = .99 \times 10^{-5} \text{ kg/m}^3 = .0099 \text{ mg/l}$$

t = 25 wks

$$P = 1.07 \times 10^{-5} \text{ kg/m}^3 = .0107 \text{ mg/l}$$

t = 40 wks

$$P = 1.0936 \times 10^{-5} \,(1-e^{-40 \times .1582}) = 1.09 \times 10^{-5} \text{ kg/m}^3 = 0.0109 \text{ mg/l}$$

Period #2 (40 to 46 Wks)

$W = 17.3$ kg/wk

$$B_s = \frac{3 \text{ mg}}{\text{m}^2/\text{day}} \times 2 \times 10^6 \text{ m}^2 \times \frac{7 \text{ day}}{\text{wk}} = 4.2 \times 10^7 \frac{\text{mg}}{\text{wk}} = \frac{42 \text{ kg}}{\text{wk}}$$

$$P = 1.09 \times 10^{-5} \text{ kg/m}^3 = 0.0109 \text{ mg/l}$$

t = 46-40 wk = 6 wks

$$P = \frac{17.3 + 42}{10^7 \times .1582}\left(1 - \exp{-6}(.1582)\right) + 1.09 \times 10^{-5} \exp{-6} \times .1582$$

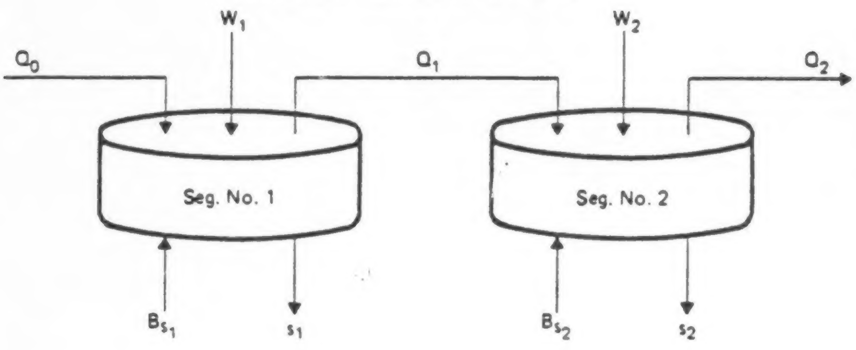
$$P = 3.75 \times 10^{-5} (1 - 0.387) + 1.09 \times 10^{-5} \times .387$$

$$= 2.3 \times 10^{-5} + .42 \times 10^{-5}$$

$$P = 2.72 \times 10^{-5} \frac{kg}{m^3} = .027 \text{ mg/l}$$

The above approach considered the lake as a single completely mixed reactor. The Great Lakes application, indicated previously, considered seven completely mixed reactors. Any type of spatial segmentation can be considered. In this case, an equation is generated for each spatial segment, and for segments that are connected by flow or dispersion, the resultant equations are coupled. Figure 3-8 contains equations and representations for several spatial segmentations which may be useful. The equation set gets complicated rather rapidly and generally requires numerical solutions using computers.

The vertically segmented system with dispersion has other potential complications in that the volumes V_1 and V_2 may change with time, and that the dispersion coefficient E_v may also change as the density gradient changes. The analysis can include a number of system features which could be important in a particular site-specific analysis. Information in the section on non-linear models is provided to assist in defining parameters when segmented systems are analyzed. It should be recognized that inclusion of these additional features does not radically change the analysis which is essentially a phosphorus residence time calculation employing lumped parameters.

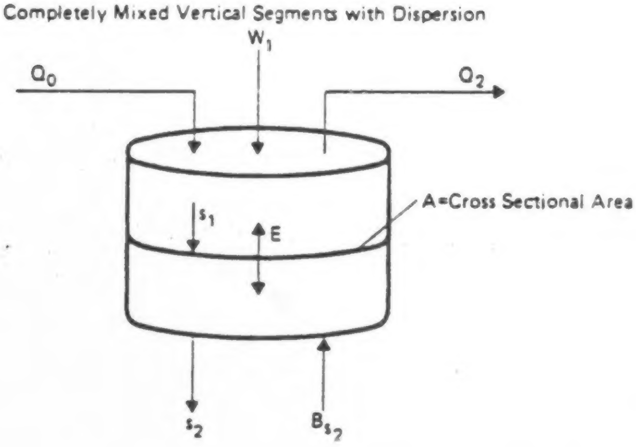


Completely Mixed Segments in Series

W_1
W_2
Q_0
Q_1
Q_2
Seg. No. 1
Seg. No. 2
B_{s_1}
s_1
B_{s_2}
s_2

$$\frac{V_1 dP_{T_1}}{dt} = W_1 - s_1 + B_{s_1} - Q_1 P_{T_1}$$

$$\frac{V_2 dP_{T_2}}{dt} = W_2 - s_2 + B_{s_2} + Q_1 P_{T_1} - Q_2 P_{T_2}$$



Completely Mixed Vertical Segments with Dispersion

W_1
Q_0
Q_2
s_1
E
A=Cross Sectional Area
s_2
B_{s_2}

$$\frac{V_1 dP_{T_1}}{dt} = W_1 - s_1 - Q_1 P_{T_1} + EA(P_{T_2} - P_{T_1})$$

$$\frac{V_2 dP_{T_2}}{dt} = -s_2 + s_1 + B_{s_2} + EA(P_{T_1} - P_{T_2})$$

Figure 3-8. Mass balance equations for horizontally or vertically separated completely mixed segments.

3.3.3 Range of Parameter Values

The values of out-flow Q, loading W, and volume V can be determined by site-specific factors as a function of time. By contrast, the values of removal rate S and source B_s are really lumped parameters which represent several phenomena. Even the first order term v is a lumped parameter which represents the settling of various types of particles and could include the resuspension of sediment due to winds. Since the parameters S, B_s, and v are lumped parameters, their value is usually determined by analysis of site-specific data. The procedures can vary from trial and error assignment of parameter values to calculations which search for the minimum of the square of the differences between calculated and observed concentrations (least squares curve fitting techniques).

The values of the lumped parameters should be either constant for the period of calculation or the allowed variations should be systematic and associated with documented phenomena such as the existence of anoxic conditions, seasonal variations in vertical structure, etc. The usual analysis considers single values of the removal parameters S and v over a yearly period. When the source term B_s is included in the analysis, it is usually varied such that B_s equals zero in the spring, winter, and fall, and has one non-zero value in the summer. The maximum variations for these parameters will be associated with an annual time scale of analysis and should consider changes in parameter values no more frequently than seasonally.

As a guide to determining the value of the lumped parameters, B_s for Shagawa Lake ranged between 6.5 to 11.3 mg/m^2 - day during anoxic periods.

This value should be very site-specific depending upon iron content and other chemical aspects of the sediments. The apparent settling velocity v' ranges from .04 and .18 m/day with a good starting value of 0.1 m/day for site-specific calculations.

The value of S has been defined (39) by:

$$S = v' A_s P_T \quad (3\text{-}36)$$

where

S = lumped settling parameter
v' = apparent settling velocity
A_s = lake surface area
P_T = total phosphorus concentration.

The value of v is defined by:

$$v = \frac{v' A_s}{V} = \frac{v'}{\bar{z}}$$

where

v = 1st order apparent phosphorus removal rate
V = lake volume
$\bar{z}$ = mean lake depth.

3.3.4 Model Calibration

The phosphorus residence time models should be calibrated by comparison of calculated and measured total phosphorus concentration data. The procedure should, if possible, employ a part of the available data base for defining parameter values and the remainder of the data base should be employed to provide a somewhat independent check on the calculations.

If the problem time scale is large with the calculation employing annual average data on flow and loads, the total phosphorus levels at turnover or in the winter should be employed to develop comparisons of calculated and observed data. For calculations on the annual time scale which employ daily or weekly averages of loads and flow, concentration data from daily or weekly sampling could be used. The data in this instance may have to be weighted by lake volume since concentration gradients are usually present during one or more seasons of the year. The calculations could be compared to surface (euphotic zone) total phosphorus values as an alternate.

The general procedure with this type of calculation framework, regardless of the time scale used, is to employ regression equations to relate total phosphorus to chlorophyll, numbers of plankton and/or dissolved oxygen which are usually the variables of concern. These curves should be lake specific and curves from the literature can be employed if some type of comparison is made between lake-specific data and results of regression analysis using data from other studies. Section 3.2 on Simplified Models provides an indication of some of the regression equations that have been developed which relate water quality variables to total phosphorus concentration.

3.3.5 Applications and Limitations of Residence Models

The discussion has centered on residence time analysis considering total phosphorus. There are no conceptual barriers to employing similar analysis for systems which are limited by other nutrients such as nitrogen or silica. Analyses considering other nutrients do not appear to have been developed or reported. Use of residence time analysis for nutrients other than phosphorus would be subjected to additional uncertainties due to the research and developmental nature of initial applications, but could be considered for use in projects where nutrients other than phosphorus appear to control eutrophication. It would be prudent to restrict these applications to sites where completed project costs are large and where extensive data on the time and space scales of concern are available, so that testing of the calculations and coefficients could be performed. Further, it will be necessary to develop site-specific information which relates the limiting nutrient concentrations to water quality variables of primary concern such as phytoplankton cell counts, chlorophyll, and/or dissolved oxygen.

The residence time analysis techniques have the advantage of providing a relatively simple framework which can be compared to observed site-specific data to analyze eutrophication in lakes. The data base required for analysis is not too extensive and can include historical information which may be available.

The disadvantage of this analysis technique is associated with the analysis of an indirect indicator of eutrophication, e.g., total nutrient concentration rather than chlorophyll or dissolved oxygen, and that lumped parameters and coefficients are employed to simulate the collective effects of a series of complex processes, such as settling, resuspension, bottom release of nutrients, etc., which can individually influence the system response to nutrient removal actions. Finally, the calculations do not include other factors such as light limitations, hydraulic limitations on growth, predation, etc.

Phosphorus residence time analysis has been applied to small lakes with moderate environmental risk and control costs. The analysis has also been employed for large lake systems with large environmental and control costs. In the case of the Great Lakes, other analysis techniques were also employed to develop information for decision making. It is suggested that consideration be given to restricting the use of this type of analysis technique to small or moderate size projects (measured by environmental risk and costs of solutions) in contrast to larger projects. This limitation is primarily associated with the absence from the analysis framework of primary variables such as dissolved oxygen and chlorophyll, and other potentially important factors such as light limitations, etc. The use of lumped parameters also provides a part of the basis for this suggestion.

3.4 NON-LINEAR EUTROPHICATION MODELING

3.4.1 General

Non-linear eutrophication modeling is an outgrowth of the pioneering work of G.A. Riley in the mid-1940's (47, 48, 49). The advent of computers and the focus of public concern on water quality issues led to expansion and application of this earlier work by O'Connor (46, 50) et al. in the late 1960's and early 1970's. There has been a rapid expansion of the processes included in analysis techniques of this type and an increase in the number of locations and water body types examined. The basic approach is very complex, and the level of experience with use of this type of calculation in decision making is small. For the most part, non-linear eutrophication modeling has been and probably should continue to be considered an area of active research with applications concentrated on complex problem settings where an extensive data base either exists or can be collected. The level of environmental risk and costs for control of eutrophication must usually be large to justify the cost of application of this type of analysis framework.

Non-linear eutrophication modeling can be divided into calculations which focus primarily on the base of the food chain and those that extend the simulation to include upper portions of the food web including fish. The present discussion will be limited to analysis techniques which concentrate on the base of the food chain, e.g., phytoplankton, and, in addition, will address water quality variables such as dissolved oxygen. The broader food chain or web modeling efforts are appropriate areas of research, but they are too speculative for use in waste load allocation projects.

Non-linear eutrophication modeling frameworks employ relatively large numbers of coefficients that describe the chemical, biochemical, and biological reactions in addition to coefficients which represent physical transport such as advection, dispersion and settling. The non-linear equations are solved numerically usually employing relatively large and complex computer programs. The calculated system responses are often difficult to understand due to the behavior of the non-linear equations, feedback of mass which control reaction velocities, and the sheer volume of numerical output generated by these computations. These difficulties, coupled with the usual high costs for data collection and analysis, suggest that it is imperative to include in the project team at least one individual who has had hands-on experience with non-linear eutrophication modeling if these techniques are to be used.

3.4.2 Formulations and Ranges of Coefficients

3.4.2.1 Spatial Segmentation and Transport. A wide range of spatial segmentation has been employed to analyze eutrophication processes in lakes. Initial efforts on a lake often involve a minimum of segmentation such as one or two completely mixed segments. For shallow lakes where light penetration extends to the lake bottom and where there is little or no vertical variation in water quality profiles or temperature, one completely mixed segment can be considered. In deeper lakes where a thermocline develops or light penetrates only a small portion of the total depth, vertical segmentation can be considered usually employing a top segment and a bottom segment.

More detailed segmentations have also been employed which separate the littoral zones and embayments from the pelagic regions of the lake. DiToro (51) has employed still more comprehensive segmentation which included vertical segments which extend into the bottom sediments of the lake. Bottom sediment analysis may prove to be a very good verification tool for lake models. The degree of segmentation employed for a site-specific project will depend upon the bathymetry and the nature of the eutrophication problem being analyzed. Generally, vertical stratification, bottom depth, light penetration, and transport will control segmentation. As as minimum, care should be exercised to provide sufficient segmentation so that depth-averaged growth rates of phytoplankton are reasonably representative of actual growth rates, and that vertical structure due to thermoclines can be simulated.

The simulations generally consider anywhere from seven to twenty state variables which can be defined by measurements such as nutrient forms, carbon, phytoplankton, dissolved oxygen, etc. The number of model coefficients could be two to three times the number of state variables with reaction rate constants usually comprising on the order of one half the number of model coefficients. Therefore, the addition of spatial segments rapidly increases the complexity and size of the input data, the computer program, and the computation time. A further problem can be associated with assimilation and display of the increased volume of model output.

Models which employ single, completely mixed segments adjacent to points of inflow and outflow present no unusual problems in routing of flows. Time variable flow balances are employed with evaporation and rainfall on the lake surface assumed to balance each other in most locations. It may be necessary to include variable lake volume in the calculations, if the system is subject to flow regulation for flood control, water supply, or navigation purposes.

For lakes where vertical segments are adjacent to points of withdrawal and inflow, there are several techniques which have been employed to distribute flow vertically (52). They generally are based on density compatibility which requires time dependent information on the temperature of water entering the system and the vertical temperature structure of the segmented lake system.

Horizontal diffusive transport coefficients for lakes range from 10^4 to 10^6 cm^2/sec. A reasonable value for the first approximation would be 10^5 cm^2/sec or could be estimated from:

$$E_H = 0.0056L^{1.3} \tag{3-38}$$

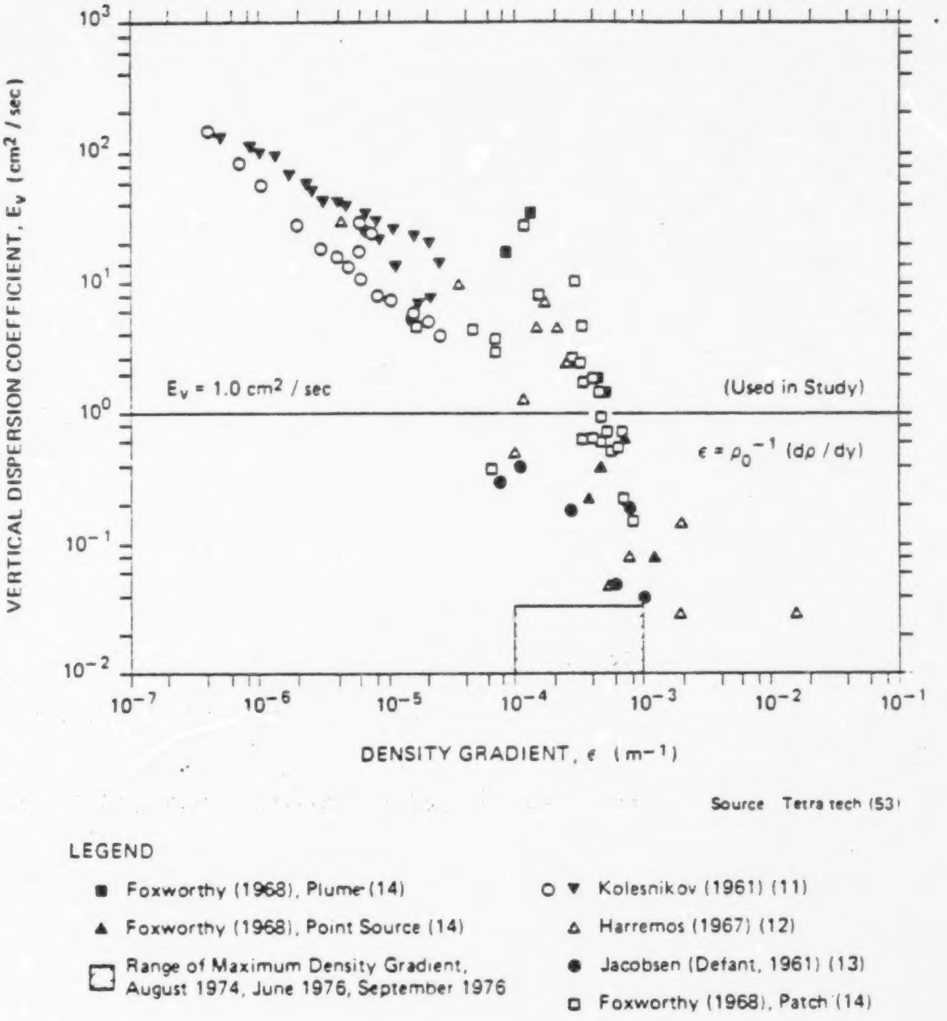
where

E_H = horizontal dispersion coefficient ($cm^2/sec.$)

L = length scale of the grid segments (cm)

The model coefficient E_H is related to the length scale of the grid employed in the computation. As this length scale exceeds 20 km, the horizontal dispersion coefficient E_H should reach a maximum constant value of 10^6 cm^2/sec.

Vertical transport usually is dominated by vertical dispersion E_V. The model parameter can be estimated from Figure 3-9 which presents data for this coefficient as a function of the density gradient. The vertical



Source Tetra tech (53)

Figure 3-9. Effect of density gradient on vertical dispersion coefficient.

dispersion coefficient will vary seasonally as the density structure of the lake changes. In addition to the data in Figure 3-8, a number of empirical formulas are available (53) to estimate this coefficient.

3.4.2.2 Kinetic Structure for Phytoplankton. The principle of conservation of mass is employed to structure the differential equations employed in non-linear eutrophication models. A mass balance around segment J which interfaces with segments k_1, k_2, ... k_n yields:

$$\frac{V_j dP_j}{dt} = \sum_{k=1}^{n} Q_{kj} P_k + \sum_{k=1}^{n} E'_{kj}(P_k - P_j) + (G_{pj} - D_{pj}) P_j V_j - S_{sj} P_j V_j + S_{sk} P_j V_j \tag{3-39}$$

where

V_j = volume of segment j

P_j = chlorophyll concentration in segment j

Q_{kj} = flows between segment k and j

P_k = phytoplankton concentration in segment k

E'_{kj} = bulk transport coefficient due to dispersion between segments k and j

G_{pj} = growth rate of phytoplankton in segment j

D_{pj} = death rate of phytoplankton in segment j

S_s = settling coefficient for phytoplankton in segment k and/or j (S_s = o for segments in the horizontal).

The first two terms on the right side of equation (3-39) represent transport, the third term incorporates growth and death, and the final terms account for settling.

Examination of this equation reveals the factors included in the analysis and, therefore, the possible factors limiting phytoplankton population levels. In lakes or segments whose detention time is small, transport can serve as an effective limitation on growth. This factor is indirectly included in the Vollenweider type analysis and in the residence time models. A second factor which can limit population levels is the growth and death terms in equation (3-39). These terms usually include, as a minimum, growth rate formulations which are temperature, nutrient, and light dependent; and death rate formulations which usually include temperature dependence, respiration, and grazing by zooplankton. These factors are not explicitly included in the less complicated methods of analysis. They are indirectly included in some aggregate form in the regressions employed to relate the limiting nutrient concentration to phytoplankton levels. The final limitation on phytoplankton levels usually included in the non-linear eutrophication models is the settling of plankton. This phenomena can be of importance from several standpoints. Settling can result in reductions in phytoplankton levels and removal of nutrients from the lake and/or from the segments of the lake where growth occurs. Settling may also create a source of oxygen demand in bottom waters below the thermocline and in the lake sediments. Discussed in previous sections, this process is included in the less complex formulations through the lumped nutrient removal parameter and indirectly in the regressions used for chlorophyll, dissolved oxygen, and sediment oxygen demands.

Growth Rate Formulation. The phytoplankton growth rate formulation can include several forms which are similar in concept but differ in the details of the formulations. The usual conceptual framework is represented by equation (3-40):

$$G_p = K_{pt}(T)r\left(\frac{C_n}{k_m + C_n}\right) \qquad (3\text{-}40)$$

where

G_p = growth rate of phytoplankton

K_{pt} = maximum specific growth rate at a reference temperature usually 20°C

(T) = temperature adjustment term

r = light-induced reduction in phytoplankton growth rate due to non-optimal incident light

k_m = Michaelis-Menton or half-saturation constant for the limiting nutrient

C_n = concentration of the limiting nutrient.

Phytoplankton growth rate G_p is usually determined by a maximum specific growth rate which is associated with a particular temperature, optimum light, and adequate nutrients. Some typical values of this coefficient are presented in Tables 3-3 and 3-4. The maximum specific growth rates used range between 0.2 and 8 per day. A starting value for the model coefficient in the range of 2 or 2.5 at 20°C could be considered in most studies.

A number of temperature formulations have been employed in models and observed for specific phytoplankton species. These formulations range from

TABLE 3-3. MAXIMUM (SATURATED) GROWTH RATES AS A FUNCTION OF TEMPERATURE

Organism	Temperature	Saturated Growth Rate, K_1 Base e, Day^{-1}
Chlorella ellipsoidea (green algae)	25 15	3.14 1.2
Nannochloris atomus (marine flagellate)	20 10	2.16 1.54
Nitzschia clostserium (marine diatom)	27 19 15.5 10	1.75 1.55 1.19 0.67
Natural association	4 2.6	0.63 0.51
Chlorella pyrenoidosa	25	1.96
Scenedesmus quadricauda	25	2.02
Chlorella pyrenoidosa	25	2.15
Chlorella vulgaris	25	1.8
Scenedesmus obliquus	25	1.52
Chlamydomonas reinhardti	25	2.64
Chlorella pyrenoidosa (synchronized culture) (high-temperature strain)	10 15 20	0.2 1.1 2.4

Source: DiToro (54).

TABLE 3-4. HALF-SATURATION CONSTANTS FOR N, P, AND Si UPTAKE (μM) REPORTED FOR MARINE AND FRESHWATER PLANKTON ALGAE (After Lehman, et al., 1975)

Cyclotella nana	NO_3	0.4-1.9	Carpenter and Guillard (1971)
		1.8	MacIsaac and Dugdale (1969)
		0.35	Caperon and Meyer (1972)
		0.5	Eppley, et al. (1969)
	NH_4	0.4	
Dunaliella tertiolecta	NO_3	0.21	Caperon and Meyer (1972)
	NH_4	0.17	
	NO_3	1.4	Eppley, et al. (1969)
	NH_4	0.6	
Asterionella japonica	NO_3	0.7-1.3	Eppley and Thomas (1969)
		1.0	Eppley, et al. (1969)
	NH_4	1.0	″
Monochrysis lutheri	NO_3	0.42	Caperon and Meyer (1972)
	NH_4	0.29	
	NO_3	0.6	Eppley, et al. (1969)
	NH_4	0.4	
Fragilaria pinnata	NO_3	0.6-1.6	Carpenter and Guillard (1971)
Bellochia sp.	NO_3	0.1-0.9	
Coccochloris stagnina	NO_3	0.31	Caperon and Meyer (1972)
Phaeodactylum tricornutum	NO_3	2.6	Ketchum (1939)
Anabaena cylindrica	NO_3	70.	Hattori (1962)
	NO_2	40.	
Chlorella pyrenoidosa	NO_2	25.	Knudsen (1965)
Chaetoceros gracilis	NO_3	0.2	Eppley, et al. (1969)
	NH_4	0.4	
Gonyaulax polyedra	NO_3	9.5	
	NH_4	5.5	
Gymnodinium splendens	NO_3	3.8	
	NH_4	1.1	
Coccolithus huxleyi	NO_3	0.1	
	NH_4	0.1	
Skeletonema costatum	NO_3	0.45	
	NH_4	0.8	
Isochrysis galbana	NO_3	0.1	
Leptocylindricus danicus	NO_3	1.25	Eppley, Rogers and McCarthy (1969)
	NH_4	0.7	
Rhizosolenia stolterfothii	NO_3	1.7	
	NH_4	0.5	
Rhizosolenia robusta	NO_3	3.0	
	NH_4	7.5	
Ditylum brightwellii	NO_3	0.6	
	NH_4	1.1	
Coscinodiscus lineatus	NO_3	2.6	
	NH_4	2.0	
Coscinodiscus wailesii	NO_3	3.6	
	NH_4	4.9	
Euglena gracilis	PO_4	16.	Blum (1966)
Cyclotella nana	PO_4	0.58	Fuhs, et al. (1972)
Thalassiosira fluviatilis	PO_4	1.72	
Chlorella pyrenoidosa	PO_4	4.-5.	Jeanjean (1969)
Nitzschia actinastreoides	PO_4	1.0	Muller (1972)
Scenedesmus sp.	PO_4	0.6	Rhee (1973)
Pediastrum duplex	PO_4	1.1	Lehman unpublished
Dinobryon cylindricum	PO_4	0.8	
D. sociale var. americanum	PO_4	0.5	
Nitzschia actinastreoides	Si	3.5	Muller (1972)
Thalassiosira pseudonana	Si	1.4-2.9	Paasche (1973a)
	Si	1.39	Paasche (1973b)
Thalassiosira decipiens	Si	3.37	
Skeletonema costatum	Si	0.80	
Licmophora sp.	Si	2.58	
Ditylum brightwellii	Si	2.96	

Source: Tetra Tech (53).

the classical formulas represented by equation (3-41) to formulations where the saturated growth rate is a maximum at some temperature and declines at lower and greater temperatures.

$$K_T = K_{20}\ \theta^{(T-20)} \qquad (3\text{-}41)$$

where

K_T = saturated growth rate at the system temperature

T = system temperature

K_{20} = saturated growth rate at the reference temperature (20°C in equation [3-41])

θ = constant whose value usually ranges between 1.01 to 1.18. A typical starting value is 1.06.

The light induced reduction in growth rate has taken several forms in the work of various investigators (49, 56). One representation, suggested by DiToro (54) and used fairly widely, is:

$$r = \frac{ef}{K_e H} \quad \exp(-a_1) - \exp(-a_o) \qquad (3\text{-}42)$$

$$a_1 = \frac{I_a}{I_s f} \exp(-K_e H) \qquad (3\text{-}43)$$

$$a_o = \frac{I_a}{I_s f} \qquad (3\text{-}44)$$

where

e = 2.71828
f = photo period
H = segment depth
K = light extinction coefficient
I_s = optimal light intensity
I_a = mean daily light intensity

Values of the light extinction can be measured or calculated depending on the complexity of the model and availability of data. The photo period and mean daily light intensity vary seasonally and can be estimated from available records. Values of the optimal light intensity I_s range between 70 to 550 Langleys(Ly)/day. A starting value of 300 Ly/day should provide an adequate point of departure for beginning calculations.

The nutrient limitations are usually formulated employing the Michaelis-Menton constant and the Monod relationship. Formulations of nutrient limitations have varied over time since the intitial modeling work in this area. There are two basic schools of thought on this issue. The first approach assumes that nutrient limitations are multiplicative. The mathematical representation of this assumption is shown in equation (3-45) for three nutrients.

$$L_n = \frac{C_p}{k_p + C_p} \quad \frac{C_n}{k_n + C_n} \quad \frac{C_s}{k_s + C_s} \qquad (3\text{-}45)$$

where

L_n = growth rate reduction factor due to all nutrient limitations

C_p = concentration of phosphorus

k_p = Michaelis-Menton constant for phosphorus

C_n = concentration of nitrogen

k_n = Michaelis-Menton constant for nitrogen

C_s = concentration of silica

k_s = Michaelis-Menton constant for silica

The second assumption that has been employed in developing formulations for the impact of limiting nutrients has utilized a Monod formulation for each nutrient which could be limiting growth.

The single nutrient limitation which results in the lowest value of L_n is then used in the calculation of growth rate G_p, for example, if

$$\frac{C_p}{k_p + C_p} < \frac{C_s}{k_s + C_s} < \frac{C_n}{k_n + C_n}$$

then:

$$L_n = \frac{C_p}{k_p + C_p} \qquad (3\text{-}46)$$

Data for the Michaelis-Menton constants are presented in Tables 3-3 to 3-7. Starting values of 25 μg N/1, 7 μg P/1, and 30 μg/1 can be considered for inorganic nitrogen, orthophosphate, and silica.

Specific Death Rate. There are several formulations which have been used or proposed for representation of the specific death rate. They generally include a respiration term k_z which is temperature corrected, and a zooplankton grazing term. A typical formulation is:

$$D_p = k_z\,(T) + C_g\left(\frac{k_{mp}}{k_{mp} + P}\right)Z \qquad (3\text{-}47)$$

Specific Death Rate = Respiration + Zooplankton Grazing

where: D_p = specific death rate

k_z = respiration rate (range .005 - .12)
(consider first estimate of .1/day)

(T) = temperature correction term

TABLE 3-5. MICHAELIS-MENTON HALF-SATURATION CONSTANTS (K_s) FOR UPTAKE OF NITRATE AND AMMONIUM BY CULTURED MARINE PHYTOPLANKTON AT 18°C. K_s UNITS ARE μMOLES/LITER (After Eppley, *et al*. 1969)

Organism	NITRATE		AMMONIUM		Cell Diameter (μ)
	K_s	± 95% Conf. Limit	K_s	±95% Conf. Limit	
Oceanic species					
Coccolithus huxleyi BT-6	0.1	0.3[b]	0.1	0.7	5
C. huxleyi F-5	0.1	1.6	0.2	0.9	5
Chaetoceros gracilis	0.3,0.1	0.5,0.2	0.5,0.3	0.5,0.3	5
Cyclotella nana 13-1	0.3,0.7	0.4,0.5	0.4	0.3	5
Neritic diatoms					
Skeletonema costatum	0.5,0.4	0.4,0.1	3.6,0.8,0.8	0.8,0.7,0.5	8
Leptocylindrus danicus	1.3,1.2	0.5,0.1	3.4,0.9,0.5	1.4,0.2,0.4	21
Rhizosolenia stolterfothii	1.7	0.4	0.5,0.5	0.9,0.4	20
R. robusta[d]	3.5,2.5	1.0,1.0	5.6,9.3	2.0,1.5	85
Ditylum brightwellii	0.6	1.7	1.1	0.6	30
Coscinodiscus lineatus	2.4,2.8	0.5,0.6	2.8,1.2	2.6,1.0	50
C. wailesii	2.1,5.1	0.3,1.8	4.3,5.5	5.4,2.0	210
Asterionella japonica	0.7,1.3	0.3,0.5	1.5,0.6	1.2,0.8	10
Neritic or littoral flagellates					
Gonyaulax polyedra	8.6,10.3	---,2.4	5.7,5.3	0.6,1.1	45
Gymnodinium splendens	3.8	0.9	1.1	1.0	47
Monochrysis lutheri	0.6	0.3	0.5	0.4	5
Isochrysis galbana	0.1,0.1	0.2,0.2	—	—	5
Dunaliella tertiolecta	1.4	1.1	0.1	0.6	8
Natural marine communities (from MacIsaac and Dugdale, 1969)					
Oligotrophic	≤0.2(6 expts)		0.1-0.6(3 expts)		
Eutrophic	≥1.0(3 expts)		1.3 (1 expt)		

---

[a] Geometric mean diameter rounded off to the nearest micron.

[b] This notation means that $0.2 < K_s < 0.4$. Negative K_s values have no physical interpretations.

[c] At 28°C, K_s for nitrate uptake was 1.0 ± 0.5; at 8°C, it was 0.0 ± 0.5.

[d] An oceanic species according to Cupp (1943).

Source: Tetra Tech (53).

TABLE 3-6. MICHAELIS-MENTON HALF-SATURATION CONSTANTS FOR NITROGEN AND PHOSPHORUS (From DiToro, *et al*., 1971)

Organism	Nutrient	Michaelis Constant, µg/Liter as N or P
Chaetoceros gracilis (marine diatom)	PO_4	25
Scenedesmus gracile	Total N Total P	150 10
Natural association	PO_4	6[a]
Microcystis aeruginosa (blue-green)	PO_4	10[a]
Phaeodactylum tricornutum	PO_4	10
Oceanic species	NO_3	1.4-7.0
Oceanic species	NH_3	1.4-5.6
Neritic diatoms	NO_3	6.3-28
Neritic diatoms	NH_3	7.0-120
Neritic or littoral Flagellates	NO_3 NH_3	8.4-130 7.0-77
Natural association Oligotrophic	NO_3 NH_3	2.8 1.4-8.4
Natural association Eutrophic	NO_3 NH_3	14 18

[a] Estimated.

Source: Tetra Tech (53).

TABLE 3-7. VALUES FOR THE HALF-SATURATION CONSTANT IN MICHAELIS-MENTON GROWTH FORMULATIONS

Phytoplankton Description	Maximum Specific Growth Rate(Days^{-1})	HALF-SATURATION CONSTANTS					Reference
		Nitrogen (mg/l)	Phosphorus (mg/l)	Silicate (mg/l)	Carbon (mg/l)	Light (Kcal/m^2/sec)	
Total Phytoplankton	0.2-8.0	0.025-0.3	0.006-0.03	-	-	-	Baca and Arnett (1976)
Total Phytoplankton	2.0	0.025	-	-	-	-	O'Connor, et al. (1975)
Total Phytoplankton	2.5	0.025	-	-	-	-	O'Connor, et al. (1975)
Total Phytoplankton	2.0	0.025	0.005	-	-	-	O'Connor, et al. (1975)
Total Phytoplankton	1.3	0.025	0.010	-	-	-	O'Connor, et al. (1975)
Total Phytoplankton	2.1	0.025	0.002	-	-	-	O'Connor, et al. (1975)
Total Phytoplankton	1.0-2.0	0.025	0.006-0.025	-	-	-	Battelle (1974)
Warm Water	2.0	0.07	0.015	-	0.03	0.002	Tetra Tech (1976)
Warm Water	1.-2	0.05-0.3	0.02-0.05	-	0.4-0.6	0.002-0.004	U.S. Army Corps of Engineers (1974)
Cold water	2.5	0.01	0.02	-	0.04	0.003	Tetra Tech (1976)
Cold Water	0.-3.	0.1-0.4	0.004-0.08	-	0.5-0.8	0.004-0.006	U.S. Army Corps of Engineers (1974)
Diatoms	2.1 (25°C)	-	-	-	-	-	Bierman (1976)
Small Diatoms	2.1	-	-	0.03	-	-	Canale, et al. (1976)
Large Diatoms	2.0	-	-	0.03	-	-	Canale, et al. (1976)
Green	1.9 (25°C)	-	-	-	-	-	Bierman (1976)
Green	1.9	0.015	0.0025	-	-	-	Canale, et al. (1976)
Blue-Green	1.6	0.015	0.0025	-	-	-	Canale, et al. (1976)
Blue-Green (N-Fixing)	0.8 (25°C)	-	-	-	-	-	Bierman (1976)
Blue-Green (Non N-Fixing)	0.8 (25°C)	-	-	-	-	-	Bierman (1976)
Small Cells Favoring Low Nutrients	1.0	0.3	0.03	-	0.5	0.003	Chen and Orlob (1975)
Small Cells Favoring Low Nutrients	1.5	0.3	0.03	-	0.5	0.002	Chen (1970)
Large Cells Favoring High Nutrients	2.0	0.4	0.05	-	0.6	0.006	Chen and Orlob (1975)
Large Cells Favoring High Nutrients	2.0	0.4	0.05	-	0.6	0.004	Chen (1970)
Readily Grazed Fast Settling	1.5	0.02	0.02	-	0.05	0.003	Chen and Wells (1975)
Not Readily Grazed Not Fast Settling	2.0	0.4	0.05	-	0.8	0.006	Chen and Wells (1975)

TABLE 3-7. VALUES FOR THE HALF-SATURATION CONSTANT IN MICHAELIS-MENTION GROWTH FORMULATIONS (concluded)

Phytoplankton Description	Saturated Light Intensity (Ft-Candles)	Chemical Composition (fraction by weight)			Temperature Tolerance Limits (°C)	Location of Study	Reference
		C	N	P			
Total Phytoplankton	-	-	-	-	-	-	Baca and Arnett (1976)
Total Phytoplankton	300	-	-	-	-	San Joaquin River	O'Connor, et al. (1975)
Total Phytoplankton	300	-	-	-	-	San Joaquin Delta Estuary	O'Connor, et al. (1975)
Total Phytoplankton	300	-	-	-	-	Potomac Estuary	O'Connor, et al. (1975)
Total Phytoplankton	350	-	-	-	-	Lake Erie	O'Connor, et al. (1975)
Total Phytoplankton	350	-	-	-	-	Lake Ontario	O'Connor, et al. (1975)
Total Phytoplankton	-	-	-	-	-	Grays Harbor/Chehalis River, Washington	Battelle (1974)
Warm Water	-	0.4	0.08	0.015	10-30	N. Fork Kings River, Calif.	Tetra Tech (1976)
Warm Water	-	-	-	-	10-30	-	U.S. Army Corps of Engineers (1974)
Cold Water	-	0.4	0.08	0.015	5-25	N. Fork Kings River, Calif.	Tetra Tech (1976)
Cold Water	-	-	-	-	5-25	-	U.S. Army Corps of Engineers (1974)
Diatoms	-	-	-	-	-	Saginaw Bay, Lake Huron	Bierman (1976)
Small Diatoms	-	-	-	-	-	Lake Michigan	Canale, et al. (1976)
Large Diatoms	-	-	-	-	-	Lake Michigan	Canale, et al. (1976)
Green	-	-	-	-	-	Saginaw Bay, Lake Huron	Bierman (1976)
Green	-	-	-	-	-	Lake Michigan	Canale, et al. (1976)
Blue-Green	-	-	-	-	-	Lake Michigan	Canale, et al. (1976)
Blue-Green (N-Fixing)	-	-	-	-	-	Saginaw Bay, Lake Huron	Bierman (1976)
Blue-Green (Non N-Fixing)	-	-	-	-	-	Saginaw Bay, Lake Huron	Bierman (1976)
Small Cells Favoring Low Nutrients	-	-	-	-	-	Lake Washington	Chen and Orlob (1975)
Small Cells Favoring Low Nutrients	-	-	-	-	-	San Francisco Bay Estuary	Chen (1970)
Large Cells Favoring High Nutrients	-	-	-	-	-	Lake Washington	Chen and Orlob (1975)
Large Cells Favoring High Nutrients	-	-	-	-	-	San Francisco Bay Estuary	Chen (1970)
Readily Grazed Fast Settling	-	0.5	0.09	0.015	-	Boise River, Idaho	Chen and Wells (1975)
Not Readily Grazed Not Fast Settling	-	0.5	0.09	0.015	-	Boise River, Idaho	Chen and Wells (1975)

Source: Tetra Tech (53).

$= \theta^{(T-20)}$ where $\theta \cong 1.08$

C_g = herbivorous zooplankton grazing rate (range 0.13 to 1.2) (consider first estimate of .25 l/mg-C-day)

Z = zooplankton carbon

K_{mp} = Michaelis-Menton half-saturation constant for zooplankton grazing on phytoplankton (range 10 - 50) (consider first estimate of 50 μg chlor/l)

P = phytoplankton (chlorophyll) concentration.

The settling term S_s in equation (3-39) can be represented by:

$$S_s = \frac{W}{H} \tag{3-48}$$

where

S_s = settling rate

H = segment depth

W = settling rate of phytoplankton (range 0 to .5) (consider first estimate of 0.1 m/day).

The non-linear eutrophication models continue the computations by simultaneously solving comparable equations for key nutrient forms, detritus carbon, zooplankton, dissolved oxygen, etc. Usual forms of these equations are:

Inorganic nutrients (for each nutrient considered):

$$\frac{dN_I}{dt} = T_{N_I} - a_1 G_p(I,T,N_I)P + a_2 R_p(T)P + a_3 R_{tz}(T)Z + a_4 K_o N_o + S_{N_I} \tag{3-49}$$

Organic nutrients (for each nutrient considered):

$$\frac{dN_o}{dt} = T_{N_o} - a_4 K_o N_o - N_o S_{ri} + a_5 R_p(T)P + a_6 R_z(T)Z + S_{N_o} \qquad (3\text{-}50)$$

Zooplankton (for each type considered):

$$\frac{dZ}{dt} = T_z + a_7 K(T)PZ - R_z(T)Z \qquad (3\text{-}51)$$

Dissolved oxygen:

$$\frac{dO_z}{dt} = T_{O_z} - K_a(C_s - O_z) + a_8 G_p(I,T,N_I)P - a_9 R_z(T)Z - a_{10} R_p(T)P$$

$$- a_{11} K_o N_o - B \qquad (3\text{-}52)$$

where

P = phytoplankton biomass (chlorophyll *a*)

t = time

G_p = phytoplankton growth rate which is defined by equation (3-40)

$R_p(T)$ = phytoplankton respiration adjusted for temperature, defined by equation (3-47)

T_n = net transport

$K(T)$ = zooplankton grazing, defined by equation (3-47) (grazing)

(Z) = zooplankton biomass

N_I = inorganic nutrient concentrations

N_o = organic nutrient concentrations

O_z = dissolved oxygen concentrations

$R_z(T)$ = zooplankton respiration and death rate, temperature corrected

K_o = decay rate (hydrolysis, mineralization, biochemical degradation) of non-living organic nutrient forms to inorganic forms

S_{N_I} = sources and other sinks of inorganic nutrients

S_{N_o} = sources and other sinks of organic nutrients

B = bottom oxygen demand

S_{ri} = settling rate of non-living particulates

K_a = reaeration coefficient

C_s = oxygen saturation value

a_1 to a_{11} = appropriate stoichiometric and yield coefficients.

3.4.3 Calibration and Verification

The non-linear eutrophication models require extensive calibration and verification. Model output calculations, generally, should be compared to data obtained over a full year and several years of data are required for proper verification. The literature contains a number of illustrations of model calibrations and verifications (56, 57, 58, 59, 24). All state variables, including the species distributions of chemicals such as orthophosphate and total phosphorus, should be employed for developing comparisons of calculated and observed water quality. Further, data from special studies such as primary productivity and bottom release rates should be employed to test the model calculations. Comparison of computed steady-state conditions with analytical solutions as well as internal program checks on conservation of mass should also be considered.

Particular attention should be directed towards annual data which provide different conditions to test model adequacy. Examples of this type of situation would be associated with data for the same annual period in two years where water quality profiles were different or with data for the same annual period where the vertical structure was different over two years.

The rather stringent verification and calibration suggested for nonlinear eutrophication models is motivated in part by their extreme complexity, but they are primarily driven by the lack of adequate understanding of the fundamental processes governing eutrophication. This primary driving force is identical for all levels of eutrophication analysis, including the Vollenweider and residence time calculations. Therefore, nonlinear eutrophication models which are properly verified and calibrated will tend to be more reliable--or will at least provide a better indication of what is known versus what is not known at a specific site--than other analysis techniques. As a consequence of the lack of understanding of the fundamental processes governing eutrophication, waste load allocations for control of eutrophication in lakes are inherently high risk exercises when compared to other waste load allocation requirements, as, for example, dissolved oxygen in streams.

3.4.4 Supplemental Calculation Procedures

Calculation Procedure for Aid in Defining the Relative Limitations Placed on Phytoplankton Growth Rates by Various Processes. The calculation procedures presented below have not been employed in past studies of eutrophication. This is pointed out to emphasize that while some insights may be obtained from the calculations, there are risks in employing the procedure since it has not been tested and is particularly sensitive to the analytical accuracy of the nutrient measurements and the system coefficients selected.

Table 3-8 presentes a summary of the components employed in non-linear eutrophication models together with estimates of the range and first starting values of some model coefficients. These data could be employed to assess the relative impact of the various processes on the growth of phytoplankton.

The following example illustrates the calculation procedure:

3.4.5 Example Problem

Data: Assume measured lake data during August are as follows:

H = 10 m (depth)

K_e = 1.3./m (extinction coefficient)

$Q_I = Q_o$ = 1000 m^3/day; P_I = 10 μg Chl a/l (incoming phytoplankton)

Volume = 10^6 m^3

Orthophosphate = 10 μg/l

$NH_3 + NO_3$ = 40 μg/l

Si = 1 mg/l

P_o = 25 μg Chl a/l (outgoing phytoplankton)

Temperature = 26°C

I_a = 475 Lys/day.

Problem: Evaluate the effect of the various factors important in non-linear eutrophication models (see Table 3-8, e.g., advective transport, light limitation, temperature limitation, nutrient

TABLE 3-8. SUMMARY OF FORMULATIONS FOR FACTORS CONSIDERED IN NON-LINEAR EUTROPHICATION MODELS AND ESTIMATES FOR RANGE AND INITIAL VALUE OF COEFFICIENTS

FACTOR	FORMULATION	COEFFICIENT RANGE	INITIAL VALUE
Advective Transport	$Q_I P_I - Q_o P$	—	—
Temperature Adjustment for Growth	$\theta^{(T-20)}$	(θ) 1.01 - 1.18	1.06
Light Limitation	$r = \frac{ef}{K_e H} [\exp(-a_1) - \exp(-a_o)]$		
	$a_1 = \frac{I_a}{I_s f} \quad \exp(-K_e H)$		
	$a_o = \frac{I_a}{I_s f}$	(I_s) 70 to 550	300 Lys/Day
Nutrient Limitation Phosphorus	$\frac{C_p}{K_p + C_p}$	(K_p) 5 to 50	7 μgP/l
Nitrogen	$\frac{C_n}{K_n + C_n}$	(K_n) 10 to 400	25 μgN/l
Silica	$\frac{C_s}{K_s + C_s}$	(K_s) —	30 μg/l
Removal			
Respiration	$K_2 \theta^{(T-20)}$	(K_2) .005 - .12 (θ) 1.04 - 1.1	.1/Day 1.08
Zooplankton Grazing	$C_g \frac{kmp}{kmp + P}$	$Z(C_g)$ 0.13 - 1.2 (kmp) 10 - 50	.25 P/ μg-C-Day 50 μgCh/l
Settling	W/H	(W) 0 to 0.5	0.1 m/day

limitation, and removal process limitation) and determine if the potential for additional phytoplankton growth exists.

Solution:

$$\text{Entering Lake} = 1000\ \frac{m^3}{day} \times \frac{10\mu g\ \text{Chl a}}{1} \times \frac{10^6\ cm^3}{m^3} \times \frac{1}{10^3\ cm^3}$$

$$= 10^7 \mu g\ \text{Chl } \underline{a}/\text{day}$$

$$\text{Leaving} = \frac{1000\ m^3}{day} \times \frac{25\ \mu g\ \text{Chl } \underline{a}}{1} \times \frac{10^6}{10^3} = 2.5 \times 10^7\ \mu g\ \text{Chl } \underline{a}/\text{day}$$

Light:

$$a_o = \frac{2(475)}{300} = 3.167$$

$$a_1 = 3.167\ \exp{-1.3} \times 10 = 7.16 \times 10^{-6}$$

$$r_1 = \frac{2.718(0.5)}{0.3(10)}\ (\exp -7.16 \times 10^{-6} - \exp{-3.167})$$

$$r_1 = .1045(1.00 - .042)$$

$$r_1 = (.1045)\ (.958) = 0.100$$

Phosphorus: $r_p = \dfrac{10}{7 + 10} = .588 \Rightarrow$ Limiting nutrient is phosphorus

Nitrogen: $r_n = \dfrac{40}{25 + 40} = 0.615$

Silica: $r_s = \dfrac{1000}{30 + 1000} = .971$

Temperature (Growth):

$$(T) = \theta^{(28-20)} = 1.06^8 = 1.594$$

Temperature (Respiration):

$$(T) = \theta^{(28-20)} = 1.088 = 1.851$$

Settling:

$$\frac{W}{H} = \frac{0.1}{10} = .01/\text{day}$$

Death:

Respiration = 0.1 x 1.851 = 0.185/day (phytoplankton)
No zooplankton data

Summary:

Transport reduces phytoplankton by:

1.5×10^7 µg Chl/day

Respiration reduces phytoplankton by:

$$0.185 \times 25\ \mu g\ \text{Chl}\ \underline{a}/1 \times 10^6\ m^3 \times \frac{10^6\ cm^{-3}}{m^3} \times \frac{1}{1000\ cm^3}$$

$= 4.6 \times 10^9$ µg Chl a/day

Settling reduces phytoplankton by:

$$0.01 \times 25 \times \frac{10^6 \times 10^6}{10^3} = .25 \times 10^9\ \mu g\ \text{Chl}\ \underline{a}/\text{day}$$

Growth increases phytoplankton by:

$$= 25 \times 1.594 \times .100 \times .588 \times 25 \times 10^6 \times \frac{10^6}{1000}$$

$= 5.7 \times 10^9$ µg Chl a/day

Based on the above calculations:

1. The influence of transport at 1000 m^3/day is small (e.g., $1.5 \times 10^7 << 4.6 \times 10^9$ (loss due to respiration)
2. Light limits growth significantly, r = 0.100.
3. Phosphorus is the limiting nutrient and is less limiting than light, $r_p = .588 > r_l = .100$.
4. The calculations indicate very little potential for additional phytoplankton growth.

The basic shortcoming of these calculations is associated with the lack of confirmation for the coefficients employed in the calculation. These computations could be employed as a rapid way of developing some indication of the importance of various processes with respect to system response.

3.4.6 Vertical Dissolved Oxygen Analysis

One of the most significant factors responsible for the vertical gradients in water quality is the density stratification due to temperature. This condition is most pronounced during the summer and generally produces a relatively well-mixed surface layer and a poorly-mixed lower layer. Differences in concentration of many water quality parameters exist between the two layers during this period and are particularly evident in the case of dissolved oxygen. Its concentration is affected not only by the vertical stratification and the associated dispersion, but also by the various sources and sinks in each zone-- photosynthetic production and exchange with the atmosphere in the upper layer, and biological respiration and benthal demand in the lower. The following analysis includes these reactions with vertical dispersive transport under a steady-state condition and has been employed in analysis of dissolved oxygen in the New York Bight.

3.4.6.1 Basic Equations and Boundary Conditions. The basic differential equation which defines the vertical distribution of dissolved oxygen under steady-state conditions is as follows:

$$0 = \frac{d}{dz} \left(E(z) \frac{dc}{dz}\right) + \sum S_o - \sum S_i \qquad (3\text{-}53)$$

in which:

c = concentration of dissolved oxygen
$E(z)$ = vertical dispersion coefficient
$\sum S_o$ = kinetic sources
$\sum S_i$ = kinetic sinks.

The concentration c may be expressed in terms of the deficit $D = c_s - c$, in which c_s equals equilibrium saturation value of dissolved oxygen for a given surface temperature. The primary kinetic source is the photosynthetic production of oxygen by phytoplankton and the sink is algal and bacterial respiration. Equation (3-53) may then be expressed as follows:

$$0 = \frac{d}{dz} \left(E(z) \frac{dD}{dz}\right) + R(z) - P(z) \qquad (3\text{-}54)$$

in which $R(z)$ and $P(z)$ are the volumetric oxygen utilization and the production rates, respectively. The former includes both the phytoplankton and bacteria contributions. Since production and respiration are operative in the surface layer, while only the latter is effective in the lower layer, the water column may be divided into two regions delineated by the thermocline.

Equation (3-54) directly applies to the upper layer, while the lower layer is described by this equation without the production term.

Since there are two second-order differential equations, one for each layer, four boundary conditions are required to evaluate the constants of integration. The upper layer is identified by the subscript T and the lower by the subscript B. The boundary conditions are provided by flux balances at the air water interface, the thermocline, and the bed, and by the concentration equality at the thermocline:

$$\text{at } z = 0: \quad E_T \frac{dD_T}{dz} = K_L D_o \tag{3-55}$$

$$\text{at } z = p: \quad D_{T_p} = D_{B_p} \tag{3-56}$$

$$E_T \frac{dD_T}{dz} = E_B \frac{dD_B}{dz} \tag{3-57}$$

$$\text{at } z = H: \quad E_B \frac{dD_B}{dz} = S \tag{3-58}$$

in which

D_o, D_p = deficits at the interface and at the pycnocline (M/L^3)

K_L = oxygen transfer coefficient (L/T)

S = areal oxygen utilization rate at the bed (M/L^2T.)

3.4.6.2. Solution of Equations. The first integration of equation (3-54) for the upper layer yields

$$E_T \frac{dD_T}{dz} = \int_o^p P(z)dz - \int_o^p R(z)dz + C_1 \tag{3-59}$$

and the second

$$D_T = \int \frac{dz}{E_T(z)} \left(\int_o^p P(z)dz - \int_o^p R(z)dz + C_1 \right) + C_2 \qquad (3\text{-}60)$$

Applying the first boundary condition (equation 3-45 to equation 3-49) yields

$C_1 = K_L D_o$

and $C_2 = D_o$

By averaging the dispersive and kinetic terms in the upper layer, equation (3-60) becomes after substituting the values of C_1 and C_2:

$$D_T = \left(\frac{P_T - R_T}{2E_T} \right) z^2 + \frac{K_L D_o z}{E_T} + D_o \qquad (3\text{-}61)$$

In the lower layer, the photosynthetic contribution is zero and the first integration of equation (3-54) yields

$$E_B \frac{dD_B}{dz} = -R_B z + C_3 \qquad (3\text{-}62)$$

Applying the fourth boundary condition (equation 3-58 to equation 3-62) provides the evaluation of C_3:

$C_3 = S + R_B H$

Substitution of this into equation (3-62) and integration leads to:

$$D_B = -\frac{R_B z^2}{2E_B} + \frac{z(S + R_B H)}{E_B} + C_4 \qquad (3\text{-}63)$$

The remaining constants D_o and C_4 are determined by the second and third boundary conditions (equations 3-56 and 3-57). Equating (3-59) and (3-62) at $z = p$ and solving for D_o yields:

$$D_o = -\frac{(P_T - R_T)}{K_L} \times p + \frac{\left[R_B(H - p) + S\right]}{K_L} \qquad (3\text{-}64)$$

Equating (3-61) and (3-63) at $z = p$ permits evaluation of C_4:

$$C_4 = +\frac{(P_T - R_T)}{2E_T} p^2 + \frac{p}{E_b}\left(\frac{R_b(P - 2H)}{2} - S\right)$$

$$+ D_o\left(1 + \frac{K_L p}{E_T}\right) \qquad (3\text{-}65)$$

Thus equations (3-61) with (3-64) define the concentration of dissolved oxygen deficit in the upper layer, and equations (3-62) with (3-65) in the lower layer. Conversion to dissolved oxygen values is made by subtracting the calculated deficit from the equilibrium saturation values specific to a given location for a given surface salinity and temperature regime.

The various transfer, kinetic and density coefficients can be assigned on the basis of either direct measurement, values reported in the literature, the previous calculations, or any combination. The following example illustrates the calculation procedure.

3.4.7 Example Problem

Data: The assumed lake parameters needed in the calculation are listed below. The values may be typical of temperate northern lakes during the summer. Values for some of the parameters listed immediately below were derived from the previous sample problem results.

Data Derived from Previous Example:

Depth (Epilimnion) = 10 m
Phytoplankton Conc. = 25 μg Chl a/1
Temperature (Epilimnion) = 26°C
r_i (light-reduction) = 0.100
Settling rate = 0.01/day

New Data:

Depth (hypolimnion) = 10 m
Temperature (hypolimnion) = 6°C
E_T [range of 10-25 ft^2/day(2)] = 2.5×10^{-1} cm^2/sec
= 23.2 ft^2/day
E_B = 1.0 m^2/day
K_L [range of 1-5 ft/day (12)] = 2.8 ft/day
= 0.864 m/day
S [range of 0.3-3 (7)] = 1 gm O_2/m^2-day
= 1 mg O_2-m/1-day

Problem: Calculate the dissolved oxygen deficit at the air-water interface, the thermocline, and at the sediment-water interface, assuming steady-state conditions.

Solution:

Deficit at the air-water interface

$$D_o = -\left(\frac{P_T - R_T}{K_L}\right) P + \frac{R_B[(H - p) + S]}{K_L}$$

where:

$P_T = P_s(r)$; r = 0.100 and p_s = 0.25 chl a; from (7)
= 0.625 mg O_2/1-day

R_T = (0.025)(chl a); from (7)

$= .625$ mg O_2/l-day

$\dot{R}_B$: No specific formulation exists for this term; however, as noted in the previous example, the rate of loss of material from the epiliminion into the hypolimnion is one-tenth the rate of loss of material due to respiration. In addition, the temperature is cooler in the epilimnion, so the rate of O_2 consumption should be reduced.

$$R_B = 0.1\ (R_T)\ (\text{T correction})$$

$$= (0.1)\ (0.625)\ 1.08^{(6-20)}$$

$= 0.0213$ mg O_2/l-day

$$D_o = \left(\frac{.625 - .625}{.864}\right) 10 + \frac{(0.0213)\ (20-10) + 1}{0.864}$$

$= 1.404$ mg/l

Deficit at the sediment-water interface ($z = 20$)

$$D_B = -\frac{(0.0213)\ (20)^2}{(2)\ 1.0} + \frac{20\ [1 + (0.0213)\ (20)]}{1.0} + 0$$

$$+ \frac{10}{1.0}\left(\frac{(0.0213)\ (-30)}{2} - 1\right) + 1.404\left(1 + \frac{(.864)\ (10)}{2.16}\right)$$

$= 18.09$ mg/l, which implies that a large potential exists in this eutrophic lake to drive the O_2 at the sediment-water interface to 0.

Deficit at the pynocline ($z = 10$)

$$D_{Tp} = \frac{(0.864)\ (10)\ (1.404)}{2.16} + 1.404$$

$= 7.02$ mg/l $\Rightarrow$ the D.O. concentration at the pynocline is equal to 1.18 mg/l.

3.5 AVAILABLE LAKE EUTROPHICATION MODELS

This section provides a partial list and brief description of available lake eutrophication models and the firms, agencies, and individuals who have supported, developed and/or applied these models. Not all of the models listed are in the public domain and other models or modifications exist. The descriptions are intended to provide the reader with an overall idea of what is available and where further information may be obtained.

The model descriptions, presented in the form of tables, (one table per model), are based on a synthesis of information from a brief questionnaire which was sent various individuals (identified in the tables as respondents) and published literature. It is hoped, therefore, that this information is reasonably current. However, modifications of these models is a continuing process and the only truly reliable sources of current information are the individuals involved in this work.

Lake eutrophication models were classified as simplified models (Section 3.2); time variable mass balance models (Section 3.3); and non-linear eutrophication models (Section 3.4). The simplified models are such that calculation can be readily performed on a calculator or programmed on a computer if desired. The programming effort is minimal and therefore packaged programs are generally not available.

Computer programs for solving the time variable phosphorus residence equations have been developed by Steven C. Chapra at the Great Lakes Environmental Research Laboratory (NOAA) and David P. Larsen at the Corvallis

Environmental Research Laboratory (EPA). Brief descriptions of each of these models are contained in Tables 3-9 and 3-10. Other computer programs may also exist, and consideration could be given to program development on a site-specific project since programming does not necessarily require a major effort.

Computer programs for solving the time-variable non-linear eutrophication equations in one, two or three dimensions represent a major development effort; therefore, the potential user should make use of available models to the extent possible. Moreover, the application and interpretation of the output from such models require an experienced analyst. Thus, in reviewing available models, the personnel who would be involved in conducting the study should be carefully considered.

The non-linear eutrophication models described herein are:

Model	Table
Water Analysis Simulation Program (WASP) (includes LAKE1A, ERIE01, and LAKE3)	3-11
WASP and Advanced Ecosystem Modeling Program (AESOP)	3-12
CLEAN Program	3-13
LAKECO, and ONTARIO	3-14
Water Quality for River Reservoir Systems (WQRRS)	3-15
Grand Traverse Bay Dynamic Model	3-16

NOTE: WASP and AESOP are related models as are LAKECO and WQRRS.

Table 3-9. DESCRIPTION OF CHAPRA'S TIME VARIABLE PHOSPHORUS MODEL

Name of Model: Time Variable Total Phosphorus Model

Respondent: Steven C. Chapra

Developer: Steven C. Chapra
Great Lakes Environmental Research Laboratory
2300 Washtemaw Ave.
Ann Arbor, Michigan 48104
(313) 668-2250

Year Developed: 1974

Capabilities: Model framework capable of computing deleterious effects of eutrophication as a function of human development of drainage basin for a series of individual lakes (developer's note).

The model considers each lake or major segments of each lake as completely mixed segments subject to waste sources, inflow and outflow, dispersion, and in-lake losses of total phosphorus. Waste sources include domestic waste, runoff from agricultural, urban and forested areas, and atmospheric fallout. (Separate algorithms are contained in the model to estimate these loads based on land use and population statistics and unit loading coefficients.) In-lake losses are estimated using the apparent settling velocity approach. The model time step used in the application to the Great Lakes was one year and the resulting projections were annual average values.

Verification: See references cited below.

Availability: Model in public domain

Applicability: The approach is general but the parameters are site specific for Great Lakes.

Support: User's Manual
There is no user's manual but several papers (see references) contain the general information needed to run the model.

Technical Assistance
Extent of technical assistance would depend on user affiliation and nature of application. At a minimum general guidance and response to questions would be provided.

References: Chapra (45,60,61)

Table 3.10. DESCRIPTION OF LARSEN'S TIME VARIABLE PHOSPHORUS MODEL

Name of Model: Phosphorus Mass Balance Model

Respondent: David P. Larsen

Developers: David P. Larsen and John Van Sickle
Corvallis Environmental Research Laboratory (CERL)
U.S. Environmental Protection Agency
200 S.W. 35th Street
Corvallis, Oregon 97330
(503) 757-4735

Year Developed: 1978

Capabilities: Imput-output model for total phosphorus (TP), time varying, to project Shagawa Lake's response to phosphorus loading reduction and to project phosphorus pattern in absence of phosphorus loading reduction (developer's note).

The model considers the lake as completely mixed, subject to external sources, inflow and outflows, net sedimentation, and an internal source of phosphorus released from sediments. External sources of total phosphorus include domestic waste, run-off, and precipitation. Step functions are used to describe the seasonal variation of the sedimentation coefficient and the sediment release rate. The time step used was one week.

Applicability: The approach is general but parameters are site specific to Shagawa Lake.

Verification: See references cited below.

Support: User's Manual
There is no user's manual

Technical Assistance
Developers could act in advisory capacity given authority from CERL.

References: Van Sickle and Larsen (62) and Larsen et al (39)

Table 3.11. DESCRIPTION OF WATER ANALYSIS SIMULATION PROGRAM

Name of Model: Water Analysis Simulation Program (WASP)*-LAKE1A, ERIE01, and LAKE3

Respondent: William L. Richardson
U.S. Environmental Protection Agency
Large Lakes Research Stations (LLRS)
9311 Groh Road,
Grosse Ile, Michigan 48138
(313) 226-7811

Developers: Robert V. Thomann, Dominic DiToro, Manhattan College, N.Y.

Year Developed: 1975 (LAKE1)
1979 (LAKE3)

Capabilities: Model is one (LAKE1) or three (LAKE3) dimensional and computes concentration of state variable in each completely mixed segment given input data for nutrient loadings, sunlight, temperature, boundary concentration, and transport coefficients. The kinetic structure includes linear and non-linear interactions between the following eight variables: phytoplankton chlorophyll, herbivorous zooplankton, carniverous zooplankton, non-living organic nitrogen (particulate plus dissolved), ammonia nitrogen, nitrate nitrogen, non-living organic phosphorus (particulate plus dissolved), and available phosphorus (usually orthophosphate). Also, a refined biochemical kinetic structure which incorporates two groups of phytoplankton, silica and revised recycle processes is available.

Verification: See references cited below.

Availability: Models are in the public domain and are available from Large Lakes Research Stations

Applicability: The model is general; however, coefficients are site specific reflecting past studies (see references)

Support: User's Manual
A user's manual titled "Water Analysis Simulation Program" (WASP) is available from Large Lake Research Stations.

Technical Assistance
Technical assistance would be provided if requested in writing through an EPA Program Office or Regional Office.

References: Thomann (63);DiToro et al (51,59)

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*The Advanced Ecosystem Model Program (AESOP) described next is a modified version of WASP

Table 3.12. DESCRIPTION OF WATER ANALYSIS SIMULATION PROGRAM AND ADVANCED ECOSYSTEM MODELING PROGRAM

Name of Model:	Water Analysis Simulation Program (WASP) Advanced Ecosystem Modeling Program (AESOP)
Respondent:	John P. St. John HydroQual, Inc. 1 Lethbridge Plaza Mahwah, N.J. 07430 (201) 529-5151
Developers:	WASP Dominic M. DiToro, James J. Fitzpatrick, John L. Mancini, Donald J. O'Conner, Robert V. Thomann (Hydroscience, Inc.) (1970) AESOP Dominic DiToro, James J. Fitzpatrick, Robert V. Thomann (Hydroscience, Inc.) (1975)
Capabilities:	The Water Quality Analysis Simulation Program, WASP, may be applied to one-, two-, and three-dimensional water bodies, and models may be structured to include linear and non-linear kinetics. Depending upon the modeling framework the user formulates, the user may choose, via input options, to input constant or time variable transport and kinetic processes, as well as point and non-point waste discharges. The Model Verification Program, MVP, may be used as an indicator of "goodness of fit" or adequacy of the model as a representation of the real world. AESOP, a modified version of WASP, includes a steady state option and an improved transport component.
Verification:	To date WASP has been applied to over twenty water resource management problems. These applications have included one-, two-and three-dimensional water bodies and a number of different physical, chemical and biological modeling frameworks, such as BOD-DO, eutrophication, and toxic substances. Applications include several of the Great Lakes, Potomac Estuary, Western Delta-Suisun Bay Area of San Francisco Bay, Upper Mississippi, and New York Harbor.
Availability:	WASP is in public domain and code is available from USEPA (Grosse Isle Laboratory and Athens Research Laboratory). AESOP is proprietary.
Applicability:	Models are general and may be applied to different types of water bodies and to a variety of water quality problems.

Table 3.12. DESCRIPTION OF WATER ANALYSIS SIMULATION PROGRAM AND ADVANCED ECOSYSTEM MODELING PROGRAM (Concluded)

Support: User's Manual
WASP and MVP documentation is available from USEPA (Grosse Isle Laboratory)
AESOP documentation is available from HydroQual.

Technical Assistance
Technical assistance of general nature from advisory to implementation (model set-up, running, calibration/verification, and analysis) available on contractual basis.

Table 3.13. DESCRIPTION OF CLEAN PROGRAMS

Name of Model: CLEAN, CLEANER, MS. CLEANER, MINI. CLEANER

Respondent: Richard A. Park
Center for Ecological Modeling
Rensselaer Polytechnic Institute
MRC-202, Troy, N.Y. 12181
(518) 270-6494

Developers: Park, O'Neill, Bloomfield, Shugart et al
Eastern Deciduous Forest Biome
International Biological Program
(RPI, ORNL, and University of Wisconsin)

Supporting Agency: Thomas O. Barnwell, Jr.
Technology Development and Application Branch
Environmental Research Laboratory
Environmental Protection Agency
Athens, Georgia 30605

Year Developed: 1973 (CLEAN)
1977 (CLEANER)
1980 (MS. CLEANER)
1981 - estimated completion date for MINI. CLEANER

Capabilities: The MINI. CLEANER package represents a complete restructuring of the Multi-Segment Comprehensive Lake Ecosystem Analyzer for Environmental Resources (MS. CLEANER) in order for it to run in a memory space of 22K bytes. The package includes a series of simulations to represent a variety of distinct environments, such as well mixed hypereutrophic lakes, stratified reservoirs, fish ponds and alpine lakes. MINI.CLEANER has been designed for optimal user application -- a turnkey system that can be used by the most inexperienced environmental technician, yet can provide the full range of interactive editing and output manipulation desired by the experienced professional. Up to 31 state variables can be represented in as many as 12 ecosystem segments simultaneously. State variables include 4 phytoplankton groups, with or without surplus intracellular nitrogen and phosphorus; 5 zooplankton groups; and 2 oxygen, and dissolved carbon dioxide. The model has a full set of readily understood commands and a machine-independent, free-format editor for efficient usage. Perturbation and sensitivity analysis can be performed easily. The model has been calibrated and is being validated. Typical output is provided for a set of test data. File and overlay structures are described for implementation on virtually any computer with at least 22K bytes of available memory.

Table 3.13. DESCRIPTION OF CLEAN PROGRAMS (CONCLUDED)

Verification: The MINI. CLEANER model is being verified with data from DeGray Lake, Arkansas; Coralville Reservoir, Iowa; Slapy Reservoir, Czechoslovakia; Ovre Heimdalsvatn, Norway; Vorderer Finstertak See, Austria; Lake Balaton, Hungary; and Lago Mergozzo, Italy. The phytoplankton/zooplankton submodels were validated for Vorderer Finstertaler See by Collins (Ecology, vol. 61, 1980, pp. 639-649).

Availability: Model are in public domain and code is available from Richard A. Park (RPI) and Thomas O. Barnwell (EPA/Athens).

Applicability: Model is general.

Support: User's Manual
A user's manual for MS. CLEANER is available from Thomas O. Barnwell, Jr.
A user's manual for MINI. CLEANER is in preparation.

Technical Assistance
Assistance may be available from the Athens Laboratory; code and initial support is available for a nominal service charge from R.P.I.; additional assistance is negotiable.

Table 3.14. DESCRIPTION OF LAKECO AND ONTARIO MODELS

Name of Model: LAKECO*, ONTARIO

Respondent: Carl W. Chen

Developers: Carl W. Chen
Tetra Tech Inc.
3746 Mount Diablo Blvd, Suite 300
Lafayette, California 94596
(415) 283-3771

(original version developed when Dr. Chen was with Water Resources Engineers)

User Developed: 1970 (original version)

Capabilities: LAKECO
Model is one dimensional (assumes lake is horizontally homogeneous) and calculates temperature, dissolved oxygen, and nutrient profiles with daily time step for several years. Four algal species, four zooplankton species, and three fish types are represented. The model evaluates the consequences of waste load reduction, sediment removal, and reaeration as remedial measures.

ONTARIO
Same as above but in three dimensions for application to Great Lakes.

Verification: The models have been applied to more than 15 lakes by Dr. Chen and to numerous other lakes by other investigators.

Availability: The model is in the public domain and the code is available from the Corps of Engineers (Hydrologic Engineering Center), EPA, and NOAA.

Applicability: General

Support: User's Manual
User's Manuals are available from Tetra Tech, Corps of Engineers, EPA, and NOAA.

Technical Assistance
Technical Assistance is available and would be negotiated on a case-by-case basis.

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*A version of LAKECO, contained in a model referred to as Water Quality for River Reservoir Systems (WQRSS) and supported by the Corps of Engineer (Hydrologic Engineering Center) is described separately.

Table 3.15. DESCRIPTION OF WATER QUALITY FOR RIVER RESERVOIR SYSTEMS

Name of Model: Water Quality for River-Reservoir Systems (WQRRS)

Respondent: Mr. R.G. Willey
Corps of Engineers
609 Second St.
Davis, California 95616
(916) 440-3292

Developers: Carl W. Chen, G.T. Orlob, W. Norton, D. Smith
Water Resources Engineers, Inc.

History: 1970 (original version of lake eutrophication model)
1978 (initial version of WQRRS package)
1980 (updated version of WQRRS)

Capabilities: See description of LAKECO in Table 3-13 (model also can consider river flow and water quality).

Verification: Chattahoochee River (Chattahoochee River Water Quality Analysis, April 1978, Hydrologic Engineer Center Project Report)

Availability: Model is in public domain and code is available from Corps.

Applicability: Model is general.

Support: User's Manual
A user's manual is available from Corps.

Technical Assistance
Advisory assistance is available to all users. Actual execution assistance is available to Federal agencies through an interagency funding agreement.

Table 3.16. DESCRIPTION OF GRAND TRAVERSE BAY DYNAMIC MODEL

Name of Model: Grand Traverse Bay Dynamic Model*

Respondent: Raymond P. Canale
LTI, Limno-Tech, Inc.
15 Research Drive
Ann Arbor, Michigan 48103
(313) 995-3131

Developers: R.P. Canale, S. Nachiappan, D.J. Hineman, and H.E. Allen

Year Developed: 1973

Capabilities: The model discretized the bay as a collection of six well-mixed cells which were arranged such that vertically well-mixed conditions were assumed throughout the bay. The water quality parameters considered were dissolved and particulate phosphorus, particulate nitrogen, dissolved organic nitrogen, ammonia, nitrate, silica, total algae and total zooplankton. Processes accounted for included transport by water motion, growth, death, decomposition, biological uptake, predation, exchange with Lake Michigan, and direct input from the Boardman River. The system of dependent governing equations are based on mass balances for the various constituents applied to each cell. The advective and dispersive transport was based on results of a separate transient vertically well-mixed numerical model.

Verification: See references below.

Availability: Model is in public domain.

Applicability: Models are developed for specific applications; however, process formulations apart from fluid transport are general.

Support: User's Manual
None

Technical Assistance
Technical assistance could be provided on a contractual basis.

References: Canale et al (64, 65), Freedman (66)

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*Limno-Tech has developed and applied a variety of models whose characteristics depend on the application. Some of these models are included in the references.

3.6 MODEL SELECTION

The discussion on model selection will be limited to comparisons among the three classes of models rather than individual models. We choose to do this because the major differences lie among the classes of models and the differences among models in the same class are relatively small. Moreover, the models in the non-linear class (in which the most variety among models exists) contain certain process formulations and interactions which are still in the research phases where they are being modified and/or refined. Thus, valid comparison of these models would be difficult (without actual testing), subject to error, and soon outdated.

Before undertaking the model selection process, one may develop a set of criteria by asking the following questions which in turn reflect an area of concern:

Technical concerns

- Does the model simulate the important processes in the prototype?
- What are the assumptions in the model, and are they consistent with the available data and the understanding of the system?
- What are the data requirements of the model, and are these data available or would a field program be required?

Information transfer and ease of understanding concern

- Are the principles and internal operations of the model easy to understand?
- Are the results of the model easily interpreted and conveyed to others who may not have a technical background?

Resource requirement concerns

- What type of personnel are needed to operate the model, and must these personnel have previous experience with the model or the class of models?
- How expensive is it to operate the model, and what are its computer requirements?

Availability concern

- Is the model available, and how convenient is it to obtain?

Acceptability concern

- What is the general acceptability of these models in the profession, and how have they performed in the past?

Technical support concern

- Is the model well-documented, and is technical support available?

Detailed discussion of what is involved in each of these concerns (or criteria) has been presented in Chapter II, Book 1, dealing with conducting waste load allocations on streams and rivers where BOD/DO is the principal water quality issue. The reader is referred to that report for this information.

We will proceed to apply these criteria to the three classes of eutrophication models and to compare the models with respect to these criteria. We will not discuss the technical criteria presented in previous sections. However, it is important to note that the steady-state completely mixed assumptions for the simplified models only provide a basis for correlating field data. Therefore, these models are essentially empirical and they are therefore limited by the range of conditions for which they prove successful, rather than those assumptions.

Table 3.17 shows how the models compare with respect to the non-technical criteria. The table indicates that if the criteria were taken as a whole, then the preferred class would be the simplified models, and the least preferred class would be the dynamic non-linear models. The key criteria affecting this outcome are th simplicity and ease of application (low resource requirements) of the simplified models. Indeed, the simplified models have experienced widespread application compared to the more complex models. Based on this comparison, it would appear that the more complex models would be selected only when technical criteria dominated the selection process. Such conditions could possibly occur under one or more of the following conditions:

- it is essential to simulate the dynamic response of the receiving water
- spatial non-uniformities in the response of the receiving water body are clearly the result of different processes or a mix of processes being important
- the resource is highly significant with respect to beneficial use and therefore the relative cost/benefit ratio of applying the dynamic non-linear models is reasonable
- the water body is complex and therefore requires a comprehensive management plan keyed to and evaluated on the basis of controlling specific processes.

This illustration of conditions under which the dynamic non-linear models may be more appropriate presupposes a philosophy that the deterministic process of applying a specific model in a technically sound manner is a preferred basis for simulating real world effects. In the case of eutrophication models there is some controversy about their current predictive capability and much less controversy that, as our understanding of the processes

Table 3.17. NON-TECHNICAL CRITERIA APPLIED TO CLASSES OF EUTROPHICATION MODELS

Criteria	Model Class		
	Simplified	Time Varying Mass Balance	Dynamic Non-Linear
Information transfer and ease of understanding	easy	easy	difficult
Resource requirements	low	moderate	high
Availability	good	good	good
Acceptability	high	moderate	moderate
Technical support	good	good	good

affecting eutrophication improves, these models will serve an ever more significant analytic function.

SECTION 4.0

DATA REQUIREMENTS

4.1 INTRODUCTION

The type, amount and quality of data used in a wasteload allocation decision where the receiving water is a lake or impoundment and where the water quality concern is potential eutrophication will depend on a number of factors. Frequently, the most important consideration is the availability of funding for the study. In some cases the physical, chemical, and biological characteristics of the lake itself are a consideration of equal or greater importance (this consideration relates directly to the type of model chosen and the model's subsequent data needs, i.e., for calibration/verification). Another consideration is the nature of the causal relationship between nutrient loads and lake response. Some general comments about the effects of these considerations on data requirements are presented below. Following these general comments more specific guidance on the sampling and analysis programs that may be required is discussed.

4.2 GENERAL CONSIDERATIONS

Availability of funding may limit the amount of new data collected for eutrophication analysis. Detailed sampling and analytical programs require relatively large amounts of funding, which may not be available for a specific project. Funding unavailability may restrict the collection of data in both large and small lakes. A large lake may be fed by a number of tributaries and may contain areas of restricted circulation, both of which may require detailed sampling to completely define the lake and its eutrophication response. The limited availability of financial resources may require that the number of stations and/or the frequency of sampling be restricted. For small lakes, funding may be so restrictive that site-specific studies might be limited to a site reconnaissance and the collection and analysis of samples from a few well-conceived stations and during critical periods. Of course, funding restrictions could limit site-specific activities to only the site reconnaissance data. In such a case the loading data would have to be developed from literature sources.

Specific physical, chemical, and biological characteristics may also determine the need for additional sampling and analytical services. On one hand, the ambient water quality of a small completely mixed lake probably could be characterized by sampling and analysis at a single station near the center of the lake. On the other hand, an elongated impoundment stretching many miles along a submerged river gorge could not be characterized by a single in-lake sample and might require several samples along the longitudinal axis of the impoundment. In addition, the choice of available modeling approaches and their different demands for data used in calibration/verification is primarily determined by lake characteristics, although economics also may play an important role in model selection. Again, a small completely mixed lake might be best modeled by a simple mass balance formulation. A large lake with many tributary, point and non-point source inputs and regions of restricted circulation would be most amenable to a more complex application of the time variable mass balance formulations or one of the non-linear eutrophication models. Obviously, the non-linear model would require a large number of measurements to achieve some desired level of reliability.

As previously discussed in the report, a good general methodology to approach eutrophication analysis is to begin analyzing the situation with literature information to obtain a preliminary understanding of the causal relationships in the lake of interest. During this process, it may be possible to reduce additional data collection and analysis activities. Such a reduction could result from at least two activities. A literature review may uncover baseline data for the lake which could reduce the need for additional data collection efforts. Furthermore, a literature review and interpretation may allow you to select and ignore less important processes and/or factors affecting the eutrophication process. For instance, a calculation of the source loadings into the lake using literature data may show that a certain tributary source is unlikely to contribute much to its nutrient budget. Consequently, it may be possible to eliminate the sampling of that tributary and to assign it the values calculated from literature sources.

4.3 SPECIFIC CONSIDERATIONS IN DETERMINING DATA REQUIREMENTS

4.3.1 Problem Identification/Description

The initial step of any waste allocation study is to define the nature and the extent of the problem. Obviously, the present and/or potential eutrophication of a specific lake or impoundment is the dominant question. Data collection and analysis can help define aspects of the eutrophication problem and provide insight to the dominant causal relationships. As eutrophication models. Obviously, the non-linear model would require a large number of measurements to achieve some desired level of reliability.

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Potential sources of existing data include (67):

- state lake classification surveys
- national eutrophication survey
- U.S. Geological Survey
- National Oceanic and Atmospheric Administration
- U.S. Soil Conservation Service
- U.S. Fish and Wildlife Service
- U.S. EPA regional offices and STORET data base
- state agencies with responsibilities corresponding to the federal agencies listed above
- county health agencies
- areawide water quality management agencies
- water and wastewater treatment plant operators
- research and educational institutions.

It may be possible to glean from all these sources all the information necessary to define the problem. Information needed to define the problem includes (67):

1. summary, analysis, and discussion of historical baseline limnological data
2. presentation, analysis, and discussion of one year of current baseline limnological data
3. trophic condition of lake
4. limiting algal nutrient

5. hydraulic budget for lake
6. phosphorus budget (and a nitrogen budget when nitrogen is limiting nutrient) for lake

Minimum requirements for one year of limnological data have been developed by the U.S. EPA as part of the Clean Lakes Program. The requirements are listed in Table 4-1. The data needs listed in Table 4-1 are comprehensive and should enable the analyst to gain some understanding of in-lake processes and to determine the trophic condition of the lake and the limiting nutrient (see Section 3.1). General methodologies for determining a nutrient budget have been previously described in this report (see Section 3.1 also). Lee and Jones (68) suggest that water samples should be collected on a weekly or no less than biweekly basis from each tributary of the water body which is expected to contribute ten percent or more of the total nitrogen or phosphorus input to the water body. These samples should be collected at a point immediately upstream of any backwater area of the water body and near a point suitable for tributary discharge measurements. In addition, they note that special measurements need to be taken during high-flow periods, as these flows often transport a substantial portion of total nutrient input. They caution that because most of these nutrients may be associated with particulate matter, the additional load of available forms of nutrients introduced during high flow may be minimal.

Reckhow (69) has recently reviewed the data on the tributary sampling frequency required to adequately define the phosphorus flux into a lake. According to his literature evaluation it would appear that a concentration

Table 4-1. BASELINE LIMNOLOGICAL MONITORING PROGRAM

1. Sampling Station Location - A single in-lake site located in an area that best represents the limnological properties of the lake, preferably the deepest point in the lake. Additional samples may be warranted in cases where lake basin morphometry creates distinctly different hydrologic and limnologic sub-basins; or where major lake tributaries adversely affect lake water quality.

2. Sampling Depth - Samples must be collected between one-half meter below the surface and one-half meter of the bottom, and must be collected at intervals of every one and one-half meters, or at six equal depth intervals, whichever number of samples is less.

3. Sampling Frequency - Sample monthly during the months of September through April and biweekly during May through August. The sampling schedule may be shifted according to seasonal differences at various latitudes. The biweekly samples must be scheduled to coincide with the period of elevated biological activity. If possible, a set of samples should be collected immediately following spring turnover of the lake.

4. Sampling Period - Samples must be collected between 0800 and 1600 hours of each sampling day unless diel studies are part of the monitoring program.

5. Chemical Constituents - All samples must be analyzed for total and soluble reactive phosphorus, nitrite, nitrate, ammonia, and organic nitrogen, pH, temperature and dissolved oxygen. Representative alkalinities should be determined.

6. Biological Constituents - Samples collected in the upper mixing zone must be analyzed for chlorophyll a. Algal biomass in the upper mixing zone should be determined through algal genera identification, and cell density counts (number of cells per milliliter), and converted to cell volume based on factors derived from total measurements, then reported in terms of biomass for each major genera identified.

7. Physical Measurements - Secchi disk depth and suspended solids must be measured at each sampling period. The surface area of the lake covered by macrophytes between zero and the ten meter depth contour or twice the Secchi disk transparency depth, whichever is less, must be reported. In addition, the surface area, the maximum depth, the average depth, the hydraulic residence time, the area of the watershed (separated into agricultural, urban, and forest) draining to the lake, the lake bathymetry and a hydraulic budget including groundwater inflow should be determined.

Source: U.S. EPA (67)

sampling interval of between 14 and 28 days is sufficient to reduce the standard error of the annual phosphorus flux to between 10 and 20 percent of the true "flux". In terms of confidence intervals at some specified level of statistical significance, we know that for sample sizes greater than six, the "t" statistic at the 95 percent confidence level varies from 2.5 to 2. Consequently the predicted range of the standard error of the mean from 10 to 20 percent indicates a range in confidence intervals at the 95 percent confidence level from 25 to 50 percent for few samples (i.e., 6 samples) to 20 to 40 percent for many samples. Reckhow offered the following comments about these conclusions, though (69):

1. More frequent sampling will still reduce uncertainty in the phosphorus concentration, but at a reduced efficiency.
2. Less frequent sampling can still be used to estimate phosphorus concentration, but at a greater risk of significant error
3. Sampling should not be systematic with respect to time (e.g., every two weeks). A better approach is to establish sampling as systematic with respect to flow, with a random start. This means that the year should be divided into n equal flow periods, for the purpose of taking n concentration samples per year.
4. Consideration should be given to a separate storm event sampling program, particularly if it is believed that a significant fraction of the phosphorus mass flux is transported during a few major events.

If the sampling program is random, i.e., in choosing a set of n observations, every possible combination of n observations should have an equal chance of being selected, then it is possible to select a statistically significant sampling frequency with a specified confidence level (70). This estimate can be made if data are available to characterize the statistical distribution of the chosen parameter (typically normal or log normal). If this data is not available, it is possible to estimate the sampling frequency if an estimate of the range of parameter values can be made.

Suppose it is desired to obtain an estimate of the mean concentration (of a normal distribution) of the desired parameter within some specified error range at a certain level of statistical significance. The specified error range, E, can be calculated:

$$E = \pm t_{\alpha} S_{\bar{x}} \tag{4-1}$$

where t_{α} denotes the student's t value for a specified α and $S_{\bar{x}}$ is the standard error of the mean as determined by:

$$S_{\bar{x}} = \sqrt{\frac{S^2}{n}\left(1 - \frac{n}{N}\right)} \tag{4-2}$$

where S^2 is the sample variance, n is the number of units sampled and N is the total number of units in the population. An equation for n can be derived from equations (4-1) and (4-2) as below:

$$n = \frac{1}{\dfrac{E^2}{t_{\alpha}^{2} S_{\bar{x}}^{2}} + \dfrac{1}{N}} \tag{4-3}$$

To obtain an estimate of n, an estimate of the population variance, S^2, must be generated. If previous data is not available, an estimate can be generated from R, the estimated concentration range:

$$S^2 = \left(\frac{R}{4}\right)^2 \tag{4-4}$$

This procedure could be used to predict the sampling frequency necessary to adequately define input as well as in-lake concentrations.

4.3.2 Model Operation (Including Calibration/Verification)

The three levels of models discussed in this report have varying data requirements for their operation. Various data requirements for these models are listed in Table 4-2. Values for some of the required data, usually determined from field work or literature reviews, are generally held constant throughout the calibration/verification stage of model operation. Morphological, hydrological, climatological, and nutrient loading data would typically fall into this category. Values for the kinetic and/or stoichiometric coefficients, on the other hand, may be adjusted during calibration studies in order that the predicted values of the state variables correspond closely with the actual values. An example of such a parameter is the net sedimentation rate coefficient used in the simple mass balance approach. Although recommended values of that parameter can be obtained in the literature (initial values for these coefficients are usually obtained from the literature), the output of model runs using this value may not accurately resemble the collected data. In the case where sufficiently large differences exist between predicted and actual values, the value of the net sedimentation rate could be adjusted so that model output more closely matches actual data. Of course, if the value of the net sedimentation rate is well beyond the range of values generally used one should review the data and analysis in an effort to explain and support the selected value(s).

Table 4-2. DATA NEEDS FOR DIFFERENT MODEL TYPES

Model Element	Model Type		
	Simple Mass Balance (S.M.B.)	Time Variable Mass Balance	Non-Linear Eutrophication Models
Morphology	Volume, Average Depth, Surface Area	Volume, Average Depth, Surface Area (possibly some bathymetric information)	Volume, Average Depth, Surface Area, Bathymetry
Hydrology	Inflow (tributaries, groundwater, precipitation) and Outflow (evaporation and discharge) (average Annual Values)	Same for S.M.B except that averaging period may depend on time-step	Same as for S.M.B except that averaging period may depend on time-step
Nutrient Loading	Tributaries, Groundwater, Precipitation, Urban Runoff, Wastewater Treatment Plant, Septic Tank Seepage (average annual values)	Same as for S.M.B. except averaging period may depend on time-step	Same as for S.M.B except that averaging period may depend on time-step
Climatology	None Required	None Required	Ambient Air and Water Temperature, Insolation, Average Wind Speed.
Limnology (in-lake processes)	(No measure of ambient limnological parameters required, but an estimate of the net sedimentation rate coefficient is required)	(No measurements of ambient limnological parameters required, but estimates of a first-order lumped settling term and a lumped internal nutrient source coefficient are required)	(Measurements for a large number of limnological parameters may be required. These could include total and soluble reactive phosphorus; nitrate, nitrite, ammonia and organic nitrogen; silica; phytoplankton; and zooplankton; carbon dioxide and oxygen concentrations. Estimates for a larg number of stoichiometric and rate coefficients may be required. Such coefficients may include light extinction coefficients, temperature coefficients, half-saturation coefficients, nutrient to chlorophyll a ratios, etc. In addition, horizontal and vertical dispersion coefficients as well as advective flow terms may need to be determined.

4.3.3 Simple Mass Balance Models. The simple mass-balance models require the least data. Those models assume that: 1) the lake is completely mixed; 2) conditions in the lake are steady-state; 3) only total nutrients (both dissolved and particulate) are important; 4) net sedimentation of nutrients occurs; and 5) in-lake phosphorus concentrations which define the boundaries between oligotrophic, mesotrophic, and eutrophic lakes can be determined. This approach results in one equation which expresses phosphorus concentration in terms of areal loading rate, net sedimentation coefficient, average depth, and the reciprocal of the hydraulic residence time. This equation is then rearranged to form an expression for total phosphorus loading rate and then evaluated at the two phosphorus concentrations chosen to define trophic state boundaries. This results in two equations defining the permissible loading at which the lake will be eutrophic, mesotrophic, or oligotrophic. The analyst then compares the measured or estimated total nutrient loading with the calculated permissible loading derived from the two equations to predict the expected trophic state of the lake. The data requirements for this process are restricted to: 1) basic physical characterization of the lake (average depth, surface area, volume, net outflow rate); 2) net sedimentation rate; 3) average total nutrient budget; and 4) allowable total nutrient boundaries defining different trophic states.

Methods for determining the physical characteristics of lakes have been discussed in this report and elsewhere (67). Possible values for the net sedimentation rates have been suggested previously in Sections 3.2 as discussed above. A check on the accuracy of any choice for this value can be made during the calibration analysis. A variety of techniques requiring field and/or literature data is available for determining the nutrient budget. Even though these simple models may require nothing more than annual average nutrient budgets, it is important to obtain accurate nutrient budget data, if possible.

Allowable total nutrient concentration criteria defining the various trophic states have been discussed previously in Section 3.2. These criteria may also be defined by using the background data collected in the problem identification/description phase to develop relationships between total nutrient concentrations and Secchi disk, dissolved oxygen, phytoplankton numbers, and chlorophyll a measurements.

4.3.4 Time Variable Mass Balance Models. The data requirements for the simplest formulation, e.g., a completely mixed lake, of the time variable model can be very similar to the simple mass balance model described above. The major difference is that the time variable models would allow and, of course, require the use of the nutrient input data on a time variable basis rather than an average annual basis. This difference may be minimal because the nutrient loading data used in the simple mass balance models must be representative and must adequately incorporate temporal variations in nutrient loading. Another difference is that the time variable model requires values for the lumped first order settling term, V, and the lumped internal source of nutrient parameter, B_s, while the simple mass balance models require only a net sedimentation term. As with the net sedimentation coefficients initial values for V and B_s can be obtained from the literature.

Values of these coefficients that are appropriate for a particular lake can then be obtained during the calibration stage of model operation.

Operationally the major benefit of the time variable mass balance model is that it allows the analyst to use time variable data in the calibration/verification phase of his analysis. That is, the analyst can use the solutions of the model as well as the appropriate parameters to calculate the total nutrient concentration on a daily, weekly, monthly, and/or any other appropriate temporal basis. The analyst will presumably collect data at least as frequently as the time steps of his model. Larger differences in the size of the data base required for the time variable or simple mass balance models can arise when the physical characteristics of the lake require it to be segmented vertically or horizontally. For example, analysis of a very deep lake may require the segmentation of the lake into several distinct layers, requiring separate physical, chemical, and biological characterization. While the differences in data requirements between the simple and the time variable mass balance models may be small, the likelihood is that more data would be collected for the time-variable model because it allows more explicit use of a larger data set in the calibration/verification phases. On the other hand, all the lake data for any one year and/or source are lumped together in the simple mass-balance formulations.

Because of the need to correlate total nutrient concentrations with other parameters of interest including dissolved oxygen, Secchi disk, chlorophyll a, and phytoplankton numbers, measurements of the parameters detailed in Table 4-1 need to be implemented.

4.3.5 Non-Linear Eutrophication Models. As previously discussed in Section 3.5 a wide variety of non-linear models have been developed and applied to a variety of situations. Most of the models deal explicitly with only the lower members of the food chain (e.g., up to zooplankton), although other models deal with members of the aquatic food chain up to and including fish. Most models have the capacity for one-, two-, or three-dimensional analysis. Because of the wide variety of models available and the diversity of options within each individual model, it is not possible to explicitly detail the data requirements of non-linear eutrophication models in general. For specific discussions of the data requirements for selected models, the reader is encouraged to examine the calibration/verification efforts listed in (56, 57, 58, 59, 24).

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